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Dioxin Transport from Contaminated Sites to Exposure Locations: A Methodology for Calculating Conversion Factors

Battelle Project Management Div., Richland, WA

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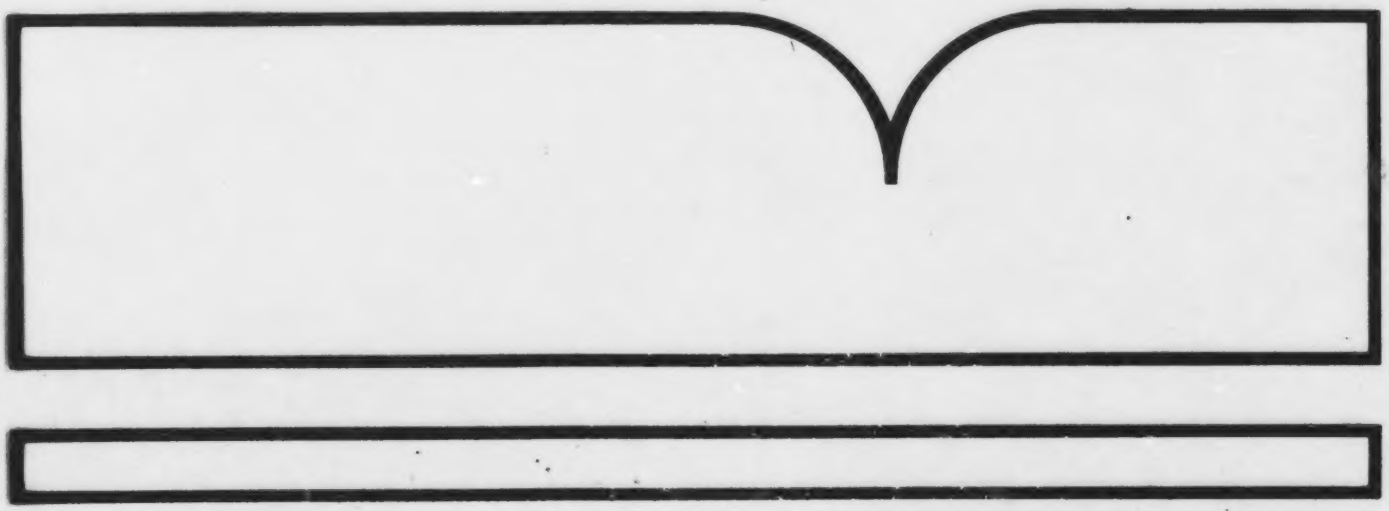

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DIOXIN TRANSPORT FROM CONTAMINATED SITES TO EXPOSURE LOCATIONS A METHODOLOGY FOR CALCULATING CONVERSION FACTORS

Prepared by

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Procedures have been developed by the US EPA for estimating the risk associated with exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin (dioxin). Concentrations of dioxin at the contaminant source are usually known, but exposure may occur at locations away from the source where concentrations are usually unknown. As a result, a need was identified for estimating dioxin concentrations away from the source.

This report discusses the transport of dioxin from a source and presents methods for estimating dioxin concentrations at potential points of exposure away from a source. The transport pathways that were considered to be important were volatilization, suspension and depositon of windblown particles, overland sediment runoff, and in-stream sediment transport. Concentrations at locations away from a source can be estimated using conversion factors for air, soil, and sediment. Concentrations in these media at potential points of exposure can be estimated using the source concentration and factors that describe the physical characteristics of the source and the transport pathways.

Because ingestion of contaminated foodstuffs will result in exposure to dioxin, an example is provided for estimating the amount of dioxin in beef. Missouri beef distribution patterns and a market dilution concept were used to estimate potential chronic exposure to contaminated beef products within the state.

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FOREWORD

The Exposure Assessment Group (EAG) of EPA's Office of Research and Development has three main functions: 1) to conduct exposure assessments; 2) to review assessments and related documents; and 3) to develop guidelines for Agency exposure assessments. The activities under each of these functions are supported by and respond to the needs of the various EPA program offices. In relation to the third function, EAG sponsors projects aimed at developing or refining techniques used in exposure assessments. This study is one of these projects and was done for the Office of Solid Waste and Emergency Response.

Dioxin problems first surfaced in the U.S. in the early 1970's with Agent Orange and the Missouri Horse Arenas. Since then, dioxin contamination has been found elsewhere in Missouri, Arkansas, Michigan, New York, and New Jersey. EPA has become increasingly involved in the discovery, assessment, and clean-up of these sites. The purpose of this document is to provide methods to use in conducting exposure and risk assessments of dioxin contamination sites.

James W. Falco, Director
Exposure Assessment Group

ABSTRACT

Procedures have been developed by the U.S. EPA for estimating the risk associated with exposure to 2,3,7,8-tetrachlorodibenzo-p-dioxin (dioxin). Concentrations of dioxin at the contaminant source are usually known, but exposure may occur at locations away from the source where concentrations are usually unknown. In response to this problem, a need was identified for estimating dioxin concentrations away from the source.

This report discusses the transport of dioxin from a source and presents methods for estimating dioxin concentrations at potential points of exposure away from a source. The transport pathways that were considered to be important were volatilization; suspension and deposition of windblown particles; overland sediment runoff; and in-stream sediment transport. Concentrations at locations away from a source can be estimated using conversion factors for air, soil, and sediment. Concentrations in these media at potential points of exposure can be estimated using the source concentration and factors that describe the physical characteristics of the source and the transport pathways.

Because ingestion of contaminated foodstuffs results in exposure to dioxin, the report includes an example of how to estimate the amount of dioxin in beef. Missouri beef distribution patterns and a market dilution concept were used to estimate potential chronic exposure to contaminated beef products within the state.

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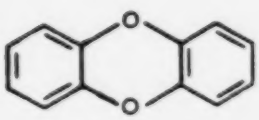
DIOXIN TRANSPORT FROM CONTAMINATED SITES TO EXPOSURE LOCATIONS:

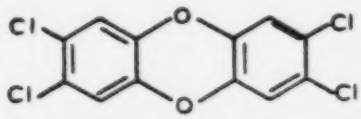
A METHODOLOGY FOR CALCULATING CONVERSION FACTORS

1.0 INTRODUCTION

A great deal of literature has been published recently concerning 2,3,7,8-tetrachlorodibenzo-p-dioxin. This compound is one extremely toxic member of a class of compounds containing the basic dioxin nucleus (Figure 1).



Dioxin Nucleus



2,3,7,8-Tetrachlorodibenzo-p-dioxin

Figure 1. Dioxin Structure

There are 75 possible chlorinated dioxins, including 22 tetrachlorodibenzo-p-dioxins. However, the 2,3,7,8-tetrachloro isomer is one of the most toxic substances known. Throughout this report, the term "dioxin" has been used to refer to the 2,3,7,8-tetrachloro isomer, the properties of which are given in Table 1. Although other isomers may be transported by the mechanisms described in this report, the other isomers have different physical properties which may render inapplicable the concentration relationships derived for the 2,3,7,8-tetrachloro isomer.

Table 1. Physical Properties of 2,3,7,8-Tetrachlorodibenzo-p-dioxin

	Esposito et al., 1980	Mabey et al., 1981	Freeman and Schroy, 1984	Perkaw et al., 1980
Molecular Weight	322	322	--	322
Melting Point, ^{o}C	305	--	--	303-305
Vapor Pressure at 25^{o}C, mm Hg	--	10^{-6}	1.5×10^{-9}	10^{-6}-10^{-7}
Water Solubility (ug/L)	0.2	0.2	0.317	0.2
Octanol-Water Partition Coefficient	--	6.9×10^{6}	--	1.38×10^{7}

In response to the discovery of a growing number of dioxin-contaminated sites, the Exposure Assessment Group within the U.S. Environmental Protection Agency Office of Research and Development has drafted procedures for estimating the human health risk associated with these sites (Schaum, 1984). The procedural algorithm developed for calculating exposure is of the form:

$$\text{Lifetime Average Exposure} = \frac{\text{Dioxin Concentration in Soil at Source} \times \text{Conversion Factor} \times \text{Contact Rate} \times \text{Exposure Rate} \times \text{Absorption Fraction}}{\text{Body Weight} \times 70 \text{ yr Lifetime}} \quad (1)$$

The algorithm contains a conversion factor that relates contamination at a point of exposure to contaminant levels at a site (e.g., conversion of a dioxin concentration in soil to a downwind airborne dioxin concentration). All conversion factors are based on dioxin concentrations in soil at the primary contaminant source. The purpose of this report is to describe and

quantify the conversion factor. The other factors in, and use of, the exposure algorithm are described by Schaum (1984).

Depending on exposure mechanisms, it is apparent that several conversion factors must be considered. Exposure to contaminants at a point away from the source may occur as a result of contaminant transport to receptors by a number of potential routes, as illustrated in Figure 2 for a generic site contaminant. Due to the intrinsic properties of dioxin, there is limited potential for transport in the dissolved phase. Thus, certain pathways in Figure 2 may be ignored. While dioxin may be transported by these routes, the concentrations at any given time would be low and would not result in high exposure risks.

The pathways of primary interest are those associated with the transport of solid particles containing adsorbed dioxin. The following mechanisms are considered to be important in the transport of dioxin from a site:

- resuspension and deposition of windblown particles;
- sediment runoff; and
- sediment transport in streams.

In addition, recent literature suggests that volatilization of dioxin from contaminated soils may occur, despite the very low vapor pressure of the pure compound (Thibodeaux, 1983; Freeman and Schroy, 1984). Consequently, vapor transport is a fourth mechanism that must be considered when evaluating the transport of dioxin from a source area.

Based on the considerations discussed above, the development of conversion factors for use in the exposure algorithm was limited to five cases as follow, where [dioxin] refers to concentration of dioxin:

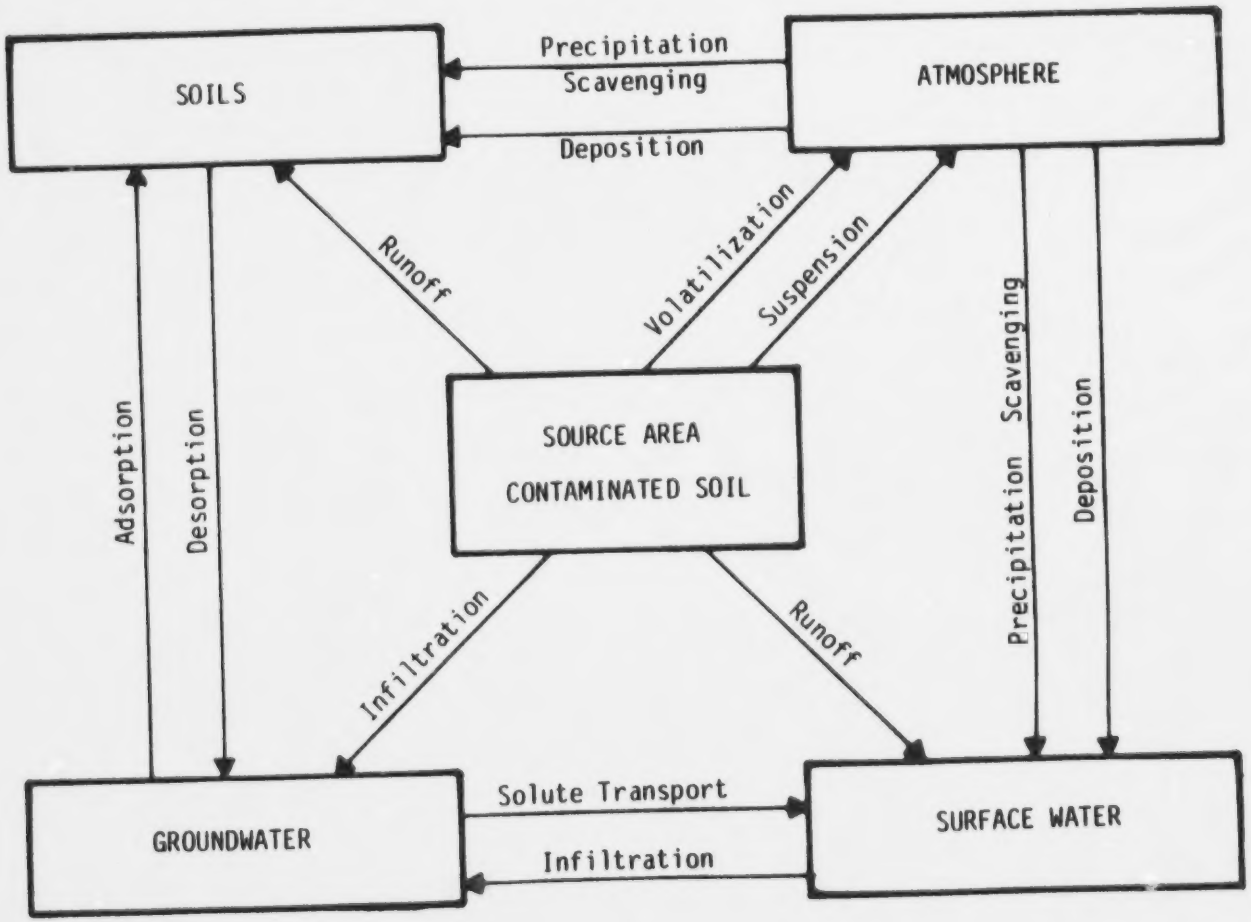


Figure 2. Pathways for Exposure from Contaminated Soils

1) $$CF_{air} = \frac{\text{[dioxin] in Air at Point of Exposure (ug/m}^3\text{)}}{\text{[dioxin] in Soil at Original Source (ug/Kg)}} \quad (2)$$

2) $$CF_{soil} = \frac{\text{[dioxin] in Soil at Point of Ingestion (ug/Kg)}}{\text{[dioxin] in Soil at Original Source (ug/Kg)}} \quad (3)$$

3) $$CF_{soil} = \frac{\text{[dioxin] in Soil at Point of Contact (ug/Kg)}}{\text{[dioxin] in Soil at Original Source (ug/Kg)}} \quad (4)$$

4) $$CF_{soil} = \frac{\text{[dioxin] in Pasture Soils (ug/Kg)}}{\text{[dioxin] in Soil at Original Source (ug/Kg)}} \quad (5)$$

5) $$CF_{sediment} = \frac{\text{[dioxin] in Sediment where Fish are Caught (ug/Kg)}}{\text{[dioxin] in Soil at Original Source (ug/Kg)}} \quad (6)$$

CF_{air} is dependent on both particulate and vapor emissions from the source. Section 3.0 describes the conversion factor for particulates and discusses recent research concerning possible vapor emissions.

The second, third, and fourth factors (CF_{soil}) differ only in the mode of exposure. The physical processes that transport contaminated soil from the source to the point of exposure are the same in all three scenarios. Thus, the CF_{soil} for all three is the same. The treatment of the different exposure routes is discussed in the predecessor paper on appropriate exposure algorithms (Schaum, 1984). The derivation of CF_{soil} is discussed in Section 4.0.

The fifth factor, $CF_{sediment}$, relates in-stream sediment concentrations to source strength. It is dependent on runoff-derived particles that accumulate in the stream bed. Estimation of $CF_{sediment}$ is discussed in Section 5.0. A summary of the three factors (CF_{air}, CF_{soil}, and $CF_{sediment}$) is provided in Section 2.0.

Each of the concentration factors presented in this report is derived from generalized ranges of environmental observations. The relationship between anthropogenic, topographic, hydrologic, climatic, and vegetative influences at a particular site will be very complex. However, consideration of these influences generally requires computer modeling, and for some influences the mathematical models do not exist.

The concentration factors developed in this report are based primarily on average, gross transport. Airborne contaminant transport is assumed to be downwind from a source. Areas of concern from overland transport will be natural drainageways and topographic breaks. Finally, catastrophic events such as hurricanes, tornadoes, and floods are not included in the derivation of conversion factors, although such events may effect significant environmental transport of contaminants. For example, Collier (1963) reported that the sediment yield from a single-day storm exceeded 40% of the yield for that year and exceeded the annual sediment yield for the previous three years. Despite the potential importance of catastrophic events in transporting contaminated material, the factors that describe the events can only be considered for specific sites. These events are not easily reduced to generic description due to their intensity, variability, and irregular occurrence. When estimation of acute event transport is necessary, ASCE (1975) should be consulted.

It must be emphasized that the conversion factor approach is a survey method to provide rapid but approximate estimates of dioxin transport and its implications. Ideally, empirical monitoring data or sophisticated numerical modeling approaches would be employed for more quantitative estimates, as

recommended by Schaum (1984). Conversion factors are useful to screen candidate source sites and to help prioritize those sites for which more quantitative estimates are needed.

The final section of this report is concerned with Missouri beef distribution patterns. The discussion in Section 6.0 is intended to provide a means for determining the extent to which Missouri inhabitants may be exposed to dioxin through consumption of contaminated beef. The emphasis is placed on developing a means of estimating the fraction of beef consumed in the state that is likely to have come from contaminated herds. Schaum (1984) has developed a method for calculating dioxin concentrations in beef as a function of soil levels in the pasture area. These concentrations are determined by using the conversion factors (CF_{soil}) derived in this work. Hence, this report complements Schaum's work by providing a complete methodology for determining, on a qualitative basis, possible dioxin exposure at sites in Missouri.

2.0 SUMMARY

Human exposure to dioxin may occur through contact with soil particles on which dioxin is adsorbed. Contaminated soil may be transported from a dioxin-contaminated source area by several mechanisms: suspension and deposition of windblown particles; overland transport and deposition of soil; and in-stream sediment transport. The relationship between conversion factors is illustrated in Figure 3.

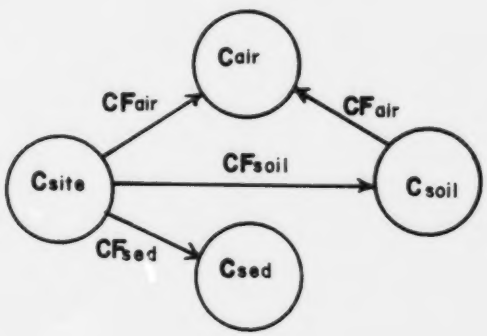


Figure 3. Relationship Between Conversion Factors

The conversion factor for atmospheric concentrations of dioxin, depending on wind speed, is determined by both the particulate and vapor levels of dioxin, as follows:

$$CF_{air} = \frac{C_{air}}{C_{soil}} = \{1 \times 10^{-7} [W^2(W - 9)]\}\left[\frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}}\right] \quad (7)$$

where C_{air} = dioxin concentration in air above soil (ug/m^3)

W = average wind velocity (m/sec)

C_{soil} = dioxin concentration in soil at point of interest (ug/Kg)

$\sigma_y\sigma_z$ = product of the Gaussian dispersion coefficients (m^2), from Figure 4

F = fetch or downwind dimension of the source (m)

x = distance from source boundary to the point of interest (m)

If $W < 10$ m/sec, the expression reduces to:

$$CF_{air} = \frac{C_{air}}{C_{soil}} = 1 \times 10^{-7} \left[\frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \right] \quad (8)$$

These expressions can be used to estimate atmospheric concentrations of dioxin, C_{air}, at the source if contaminated source conditions are used for C_{soil}. To determine atmospheric concentrations downwind from the source, soil concentrations, either calculated from conversion factors or measured, are used for C_{soil}. If C_{source} is used for C_{soil}, the degradation and dispersion factors become unity.

Dioxin concentration in the surface soil in the vicinity of a contaminated source is a function of the amount of sediment delivered to the point of exposure. The conversion factor for overland transport is:

$$CF_{soil} = \frac{C_{soil}}{C_{source}} = \frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} \quad (9)$$

where C_{soil} = dioxin concentration in soil at point of interest (ug/Kg)

C_{source} = dioxin concentration in soil at the source (ug/Kg)

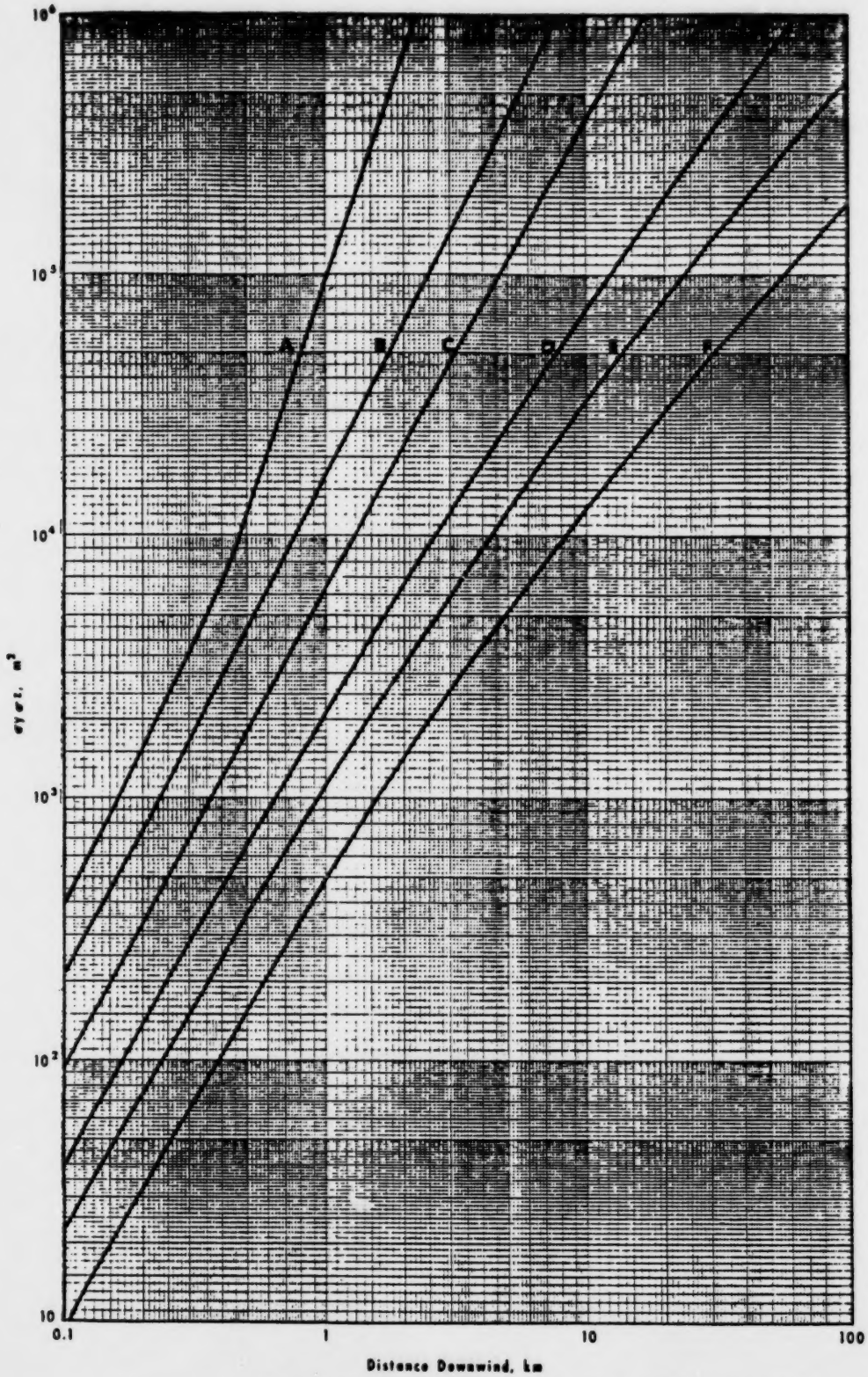


Figure 4. The Product of $\sigma_y\sigma_z$ as a Function of Downwind Distance from the Source (from Turner, 1970) for each of the Six Stability Classes A-F

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L_{source} = estimated soil loss from the Universal Soil Loss Equation (USLE) for the source (tons/acre/yr)

A_{source} = source area (acres)

L_{basin} = estimated soil loss from the USLE for the watershed upstream of the point of interest (tons/acre/yr)

A_{basin} = watershed area upstream of the point of interest (acres)

The conversion factor for windblown contributions to soil contamination is:

$$CF_{soil} = \frac{C_{soil}}{C_{source}} = \frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \qquad (10)$$

where $\sigma_y\sigma_z$ = product of the Gaussian dispersion coefficients (m^2), from Figure 4

x = distance from source boundary to the point of interest (m)

F = fetch or downwind dimension of the source (m)

If runoff patterns and wind direction are coincident, the contributions to soil contamination are additive and the expression for the conversion factor becomes:

$$CF_{soil} = \frac{C_{soil}}{C_{source}} = \frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} + \frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \qquad (11)$$

In order to determine the dioxin concentration on sediment delivered to a stream in the vicinity of a contaminated source, the procedure is the same as that used to determine surface soil concentrations. The conversion factor for estimating stream sediment concentrations at distances downstream from the source is:

$$CF_{sediment} = \frac{C_{sediment}}{C_{source}} = \frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} \qquad (12)$$

where $C_{sediment}$ = dioxin concentration in sediment at the point of interest (ug/Kg)

L_{source} = estimated USLE soil loss for source (tons/acre/yr)

A_{source} = source area (acres)

L_{basin} = estimated soil loss from the USLE for the watershed upstream of point of interest (tons/acre/yr)

A_{basin} = watershed area upstream of the point of interest (acres)

The conversion factors developed for the three exposure modes have limitations. Because chronic, long-term exposure rates are of concern, the methods developed utilize average or estimated values for parameters such as wind velocity, precipitation, runoff, soil erodibility, topography, vegetative cover, and stream characteristics. Because of these generalizations, the methods must be applied with caution, particularly in areas such as the western United States where little experimental work on sediment yields has been conducted.

Use of conversion factors is further limited by the absence of actual, comprehensive site data with which to test the methods developed by this study. Further site studies, designed to collect the necessary data on dioxin concentrations and environmental characteristics at a source and at potential points of exposure, will be necessary in order to verify these methods.

3.0 ATMOSPHERIC CONCENTRATIONS

3.1 Introduction

Human exposure to dioxin may result from inhalation of contaminated soil particles suspended in the atmosphere or vapors resulting from volatilization from contaminated soil. In order to quantify exposure, it is necessary to derive a means of estimating atmospheric dioxin levels in the environs of a contaminated source.

3.2 Particulate Exposure

Particulate exposure results from the presence of dioxin-contaminated soil in the atmosphere. Suspension of soil particles in the atmosphere may result from the erosive action of wind or from activities which disturb the soil, such as plowing or excavation. The concentration of particles in the air column and their residence time are highly dependent on particle size and atmospheric conditions. Ideally, the conversion factors should be derived from studies in which the soil size fractions were isolated, and in which the concentration versus size fraction in the air column was known at various distances from a source. Unfortunately, no data were found on particle size distribution of dioxin-contaminated soils. However, data do exist for dioxin concentrations on fly ash and larger ash particles caught in precipitators (Fred C. Hart, 1984). These data indicate that dioxin levels may increase by a factor of 2 to 12 on smaller size particles compared to larger particles.

If this relationship were found to be true for soils, it would provide some guidance for adjusting airborne particulate level estimates based on soil conditions. In the absence of data to confirm increased concentrations on small soil particles, selection of exposure factors has been based on the assumption that concentrations of dioxin adsorbed on suspended particles will be the same as dioxin concentrations in the bulk soil source. This assumption, in turn, assumes that dioxin will be adsorbed on smaller-size particles, which are subject to resuspension and respiration, at levels comparable to concentrations in bulk soil. The assumptions conform to work by Thibodeaux (1983), who found dioxin levels on dust in the air at the Vertac site to be 1.1 ug/Kg compared to soil levels of 1.3 ug/Kg.

Conversion factors can be derived by deterministic or empirical approaches. Deterministic approaches use numerical models to simulate the basic phenomena involved. The models mathematically describe the physical processes that effect transport. Many models have been developed to describe resuspension and deposition; however, inadequate data are available to validate these models (Sehmel, 1980). For those models that do exist, the large number of input requirements can be severely limiting. Sehmel (1980) lists over 40 factors which influence resuspension, although the relationships between these factors are not thoroughly understood. This degree of complexity is too great for derivation of simple site characteristic guides for predicting atmospheric contaminant or particulate levels. Gillette (1973) employed a simplified relationship to describe horizontal flux (Fh) of particulates:

$$Fh = Ch\ W_*^2(W_*-Wt) \qquad (13)$$

where W_* = wind shear velocity (m/sec)

Wt = a threshold velocity (m/sec)

Ch = empirical constant

However, because Ch is empirically derived, site-specific data are required to calibrate the algorithm.

The empirical approach is based on correlation analysis of data from actual sites, with subsequent selection of a factor that best matches the relationship between soil contamination and atmospheric contamination levels. As noted previously, this approach is required to calibrate Gillette's simplified model. A single datum has been found for atmospheric levels of dioxin at, near, or downwind from contaminated sites. Thibodeaux (1983) reviewed monitoring data from the Vertac site and found atmospheric dust concentrations of 54 ug/m^3, dioxin concentrations on the dust of 1.1 ug/Kg, and soil dioxin levels of 1.3 ug/Kg over a 37 day period. These concentrations yield a CF_{air} of 4.6 x 10^{-8} Kg/m^3. Based on only this datum, an empirical approach specific to dioxin cannot be determined at this time. However, data are available on the relationship of particulate contaminant levels and surface contaminant levels for a range of wind conditions and for a variety of soil disturbing activities. These values are summarized in Figure 5. The resuspension factors in Figure 5 were determined by comparing atmospheric contamination (volumetric) to surface soil contamination (areal) for particulate-based contaminants. The areal measurements were possible

REFERENCE	LOCATION	STRESS
STEWART, 1967	MARALINGA TRIALS	STIRRED DUST
STEWART, 1967	MARALINGA TRIALS	VEHICLE, 0.3 m
STEWART, 1967	MARALINGA TRIALS	WALKING
LANGHAM, 1971	NEVADA TEST SITE	VEHICULAR
STEWART, 1967	MARALINGA TRIALS	CAB, LANDROVER
STEWART, 1967	MONTE BELLO ISLANDS	VEHICLE, 7TH DAY
STEWART, 1967	MONTE BELLO ISLANDS	VEHICLE, 4TH DAY
CALC. FROM MILHAM et al., 1976	FIELD	MOWING
CALC. FROM MILHAM et al., 1976	FIELD	PLANTING, DISKING
CALC. FROM MILHAM et al., 1976	FIELD	SUBSOILING
STEWART, 1967	MONTE BELLO ISLANDS	VEHICLE, 7TH DAY
STEWART, 1967	C. D. TRIALS	WORK, OPEN
STEWART, 1967	AUSTRALIAN DESERT	VEHICLE TAILBOARD
STEWART, 1967	MARALINGA TRIALS	VEHICLE, 1-2 DAY
CALC. FROM BENNETT, 1976	NEW YORK	FALLOUT CONCS.
CALC. FROM MILHAM et al., 1976	FIELD	TRACTOR, DOWNWIND
CALC. FROM IRANZO AND SALVADOR, 1970	PALMARES, SPAIN	YEARLY FARMING
STEWART, 1967	AUSTRALIAN DESERT	WALKING
CALC. FROM MILHAM et al., 1976	FIELD	TRACTOR CAB
MILHAM et al., 1976	FIELD	TRACTOR
CALC. FROM MYERS et al., 1976	SLUDGE	ROTOTILLING
CALC. FROM MILHAM et al., 1976	FIELD	FERTILIZING
STEWART, 1967	MARALINGA TRIALS	WIND
ANSPAUGH et al., 1970	NEVADA TEST SITE	WIND
STEWART, 1967	SANDY-GRASS	WIND
STEWART, 1967	SANDY-DEBRIS	WIND
STEWART, 1967	MONTE BELLO ISLANDS	WIND
SEHMEL AND ORGILL, 1973	ROCKY FLATS	WIND
CALC. FROM IRANZO AND SALVADOR, 1970	PALMARES, SPAIN	WIND
LANGHAM, 1971	NEVADA TEST SITE	WIND
STEWART, 1967	PAVING STONES	WIND
CALC. FROM BENNETT, 1976	NEW YORK-FALLOUT	WIND
STEWART, 1967	SANDY-CLEARED	WIND
SEHMEL AND LLOYD, 1975	HANFORD	WIND
CALC. BY BENNETT, 1976	NEW YORK-U	WIND
HAMILTON CALC. BY BENNETT, 1976	UNITED KINGDOM-U	WIND
KREY et al., 1976	ROCKY FLATS	WIND
ANSPAUGH et al., 1975	NEVADA TEST SITE	WIND

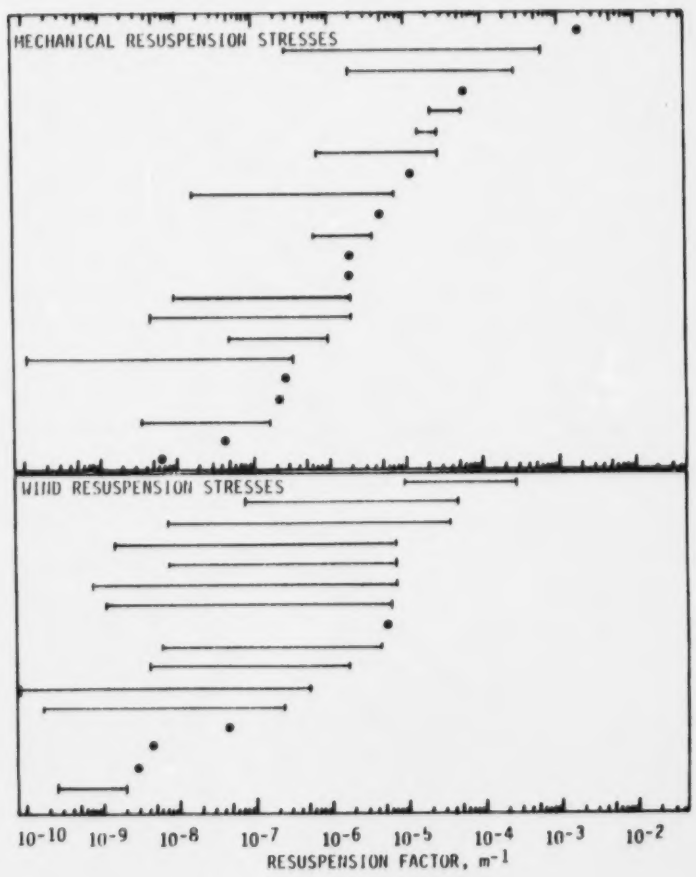

Figure 5. Relative Resuspension Factors* Under Various Site Conditions (After Sehmel, 1980)

*Resuspension Factor (RF) = Ratio of Contaminant per Volume Air to Contaminant Per Unit Area of Soil (m^{-1} units)

because radio-contaminant levels can be measured in situ without regard to sample depth. For chemicals such as dioxin, concentrations would be required.

An alternate method of calculating particulate levels in the atmosphere is to compare atmospheric monitoring data to average soil levels for conservative contaminants other than dioxin. Data were collected on fluorides, chromium, copper, manganese, nickel, vanadium, and arsenic for this approach (National Research Council, 1971; 1974a; 1973; 1975; 1974b; Versar, 1976; Sullivan, 1969; and Nriagu, 1979, 1980). Atmospheric values from rural areas were used to minimize the influence of anthropogenic materials emitted from stacks. Values for each contaminant were combined to derive a conversion factor, CF_x, which is defined as the ratio of atmospheric concentrations (ug/m^3) to soil levels (ug/Kg). Assuming atmospheric particulates are derived from nearby soils, the predicted average contaminant level in ambient air at the site can be obtained by multiplying CF_x by the average concentration of the contaminant in soil. Additionally, CF_x is an estimate of particulate levels in air (Kg/m^3). The values for CF_x determined for each of the seven contaminants are provided below:

$$\text{Fluorides: } CF_F = \frac{0.02 \text{ to } 0.05 \text{ ug F/m}^3 \text{ air}}{20{,}000 \text{ to } 500{,}000 \text{ ug F/Kg soil}} \qquad (14)$$

$$= 3 \times 10^{-6} \text{ to } 4 \times 10^{-8} \text{ Kg soil/m}^3 \text{ air}$$

(Data from National Research Council, 1971)

Chromium: $$CF_{Cr} = \frac{0.01 \text{ ug Cr/m}^3 \text{ air}}{37{,}000 \text{ ug Cr/Kg soil}} \quad (15)$$

$$= 3 \times 10^{-7} \text{ Kg soil/m}^3 \text{ air}$$

(Data from National Research Council, 1974a)

Copper: $$CF_{Cu} = \frac{0.005 \text{ to } 0.05 \text{ ug Cu/m}^3 \text{ air}}{20{,}000 \text{ ug Cu/Kg soil}} \quad (16)$$

$$= 3 \times 10^{-6} \text{ to } 3 \times 10^{-7} \text{ Kg soil/m}^3 \text{ air}$$

(Data from Nriagu, 1979)

Manganese: $$CF_{Mn} = \frac{0.08 \text{ ug Mn/m}^3 \text{ air}}{800{,}000 \text{ ug Mn/Kg soil}} \quad (17)$$

$$= 1 \times 10^{-7} \text{ Kg soil/m}^3 \text{ air}$$

(Data from National Research Council, 1973)

Nickel: $$CF_{Ni} = \frac{0.006 \text{ ug Ni/m}^3 \text{ air}}{30{,}000 \text{ to } 80{,}000 \text{ ug Ni/Kg soil}} \quad (18)$$

$$= 2 \times 10^{-7} \text{ to } 8 \times 10^{-8} \text{ Kg soil/m}^3 \text{ air}$$

(Data from National Research Council 1975 and Nriagu, 1980)

Vanadium: $$CF_{V} = \frac{0.002 \text{ ug V/m}^3 \text{ air}}{200{,}000 \text{ ug V/Kg soil}} \quad (19)$$

$$= 1 \times 10^{-7} \text{ Kg soil/m}^3 \text{ air}$$

(Data from National Research Council, 1974b)

Arsenic: $CF_{As} = \frac{0.001 \text{ ug As/m}^3 \text{ air}}{5{,}000 \text{ ug As/Kg soil}}$ (20)

$= 2 \times 10^{-7}$ Kg soil/m^3 air

(Data from Versar, 1976 and Sullivan, 1969)

In general, CF_x values are in the range of 1-3 x 10^{-7} Kg/m^3. This range is comparable to 100 ug/m^3, a level commonly found in polluted air and in excess of the Federal Ambient Air Quality Standard of 75 ug/m^3. If a typical surface soil density of 1600 Kg/m^3 and soil depth of 1.0 cm are assumed, these CF_x values would convert to resuspension factors (RF) of 1 x 10^{-7} Kg/m^3 x 1.6 x 10^{-3} m^3/Kg x 1 x 10^2 m^{-1} = 2 x 10^{-8} m^{-1}. This value is consistent with the work from which the values of RF were derived; in his work, Sehmel (1980) determined that the tracer had mixed to a depth of 1.0 cm. The value of 2 x 10^{-8} m^{-1} corresponds to the RF values in Figure 5 for the lowest wind or activity stresses. Hence, the CF_x values derived from metal concentration ratios agree with empirical data for particulates in general. Then, as a rule of thumb: C_{air} in ug/m^3 = (1 x 10^{-7}) C_{soil} in ug/Kg. This relationship compares favorably with the 4.6 x 10^{-8} Kg/m^3 value calculated from the Thibodeaux (1983) data for the Vertac site. The relationship represents average conditions over a year rather than those that would prevail during storm events or in areas with high soil disruption activity levels. The relationship addresses atmospheric levels at the perimeter of the source. For downwind concentrations, the dispersion factor developed in Sections 3.3 can be used, or CF_{air} can be calculated based on soil dioxin concentrations at the point of exposure (measured or calculated using CF_{soil} from Section 4.0).

It has been observed that soil erosion is a function of wind velocity cubed (Sehmel, 1980). More specifically, erosion is proportional to $W^2(W - W_t)$ (Gillette, 1973) where W is wind velocity and W_t is a threshold value of 6 - 13 m/sec. For calculations in wind, the median value (9) for the threshold wind velocity was used and the relationship for determining particulate concentration in air can be described as follows:

$$W < 10 \text{ m/sec}: \quad C_{air} = 1 \times 10^{-7} \, C_{soil} \tag{21}$$

$$CF_{air} = 1 \times 10^{-7} \text{ Kg/m}^3$$

$$W \geq 10 \text{ m/sec}: \quad C_{air} = (1 \times 10^{-7}) \, W^2 \, (W - 9) \, C_{soil} \tag{22}$$

$$CF_{air} = (1 \times 10^{-7}) \, W^2 \, (W - 9) \text{ Kg/m}^3$$

where C_{air} = dioxin concentration in air above soil (ug/m^3)

C_{soil} = dioxin concentration in soil at the point of interest (ug/Kg)

Because W is an average wind speed over time for chronic exposure, the second relationship will not be required unless acute exposure calculations are desired.

Mechanical disturbances can increase dust emissions significantly. For dioxin levels that would occur during episodes of mechanical disturbance, the relevant values in Figure 5 should be employed to increase CF_{air} proportionally to the ratio of the relative resuspension factors for the disturbed state and the calm state ($RF = 10^{-8} \, m^{-1}$), respectively. Hence, for

tractor use (RF = 10^{-7} to 10^{-6} m^{-1}), CF_{air} would be 1 to 2 orders of magnitude higher, or CF_{air} = 1 x 10^{-5} to 1 x 10^{-6} Kg/m^3.

This approach is based on calculation of atmospheric levels as a function of the soil at the point of interest and does not account for dispersion downwind. Thus, for downwind areas, C_{soil} is not the original concentration at the source but the concentration at the point of exposure. If a measured value is not available, an estimated value is employed using the methods described in Section 4.0. This approach was taken because atmospheric levels are believed to arise primarily from resuspension of local soils (Sehmel, 1980). This assumption further assumes that activity levels at the point of exposure are greater than or equal to activity levels at the source. More complex methods for rapid assessment of particulate emissions have been summarized by MRI (1984) and are recommended if time and resources allow.

Sites downwind from the original source will reflect dioxin levels in soil resulting from all transport mechanisms, runoff and wind. Further, atmospheric levels for chronic exposure will be a function of average wind conditions, i.e., wind speeds of about 5.5 m/sec (12 mi/hr) (Versar, 1983). Most wind-transported dioxin will have arrived from discrete storm events, therefore, the conversion factor CF_{air} for $W \geq 10$ m/sec is required to determine downwind soil levels as described in Section 4.0. These atmospheric deposition contributions are combined with runoff input, also described in Section 4.0, to yield an estimate of total average soil concentration at the downwind point of exposure. The conversion factor CF_{air} is then applied to calculate atmospheric levels, as follows:

$$C_{air} = (CF_{air})\ (C_{soil}) \qquad (23)$$

where C_{air} = dioxin concentration in air above soil (ug/m^3)

C_{soil} = dioxin concentration in soil at the point of interest (ug/Kg)

3.3 Gaussian Dispersion

The conversion factor derivation described in Section 3.2 applies only to the air directly above a point of interest. Atmospheric concentrations downwind from a source can be estimated without knowing soil concentrations if the airborne particulate plume is considered to be subject to Gaussian dispersion. In this case, atmospheric concentrations, C_{air}, at point x can be calculated using the following formula (Turner, 1970):

$$C_{air(x,o,o)} = \frac{Q_z}{\pi \sigma_y \sigma_z W} \qquad (24)$$

where C_{air} = concentration in air (g/m^3)

Q_z = emission rate (g/sec)

$\sigma_y\sigma_z$ = product of the Gaussian dispersion coefficients (m^2), from Figure 4

W = wind speed (m/sec)

Therefore, the ratio of concentrations at two points can be determined by the ratio:

$$\frac{C_{air1}}{C_{air2}} = \frac{(\sigma_y\sigma_z)_2}{(\sigma_y\sigma_z)_1} \qquad (25)$$

If C_{air2} is set at the boundary of the source, it can be equated to C_F. Therefore,

$$\frac{C_{x+F}}{C_F} = \frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \tag{26}$$

where x = distance from source boundary to the point of interest

F = fetch or downwind dimension of the source

and $$C_{x+F} = C_F \frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \tag{27}$$

The approach described by equations 24, 25, 26, and 27 assumes conservation of mass.

Values for $\sigma_y\sigma_z$ downwind from a source area are shown in Figure 4 for the following six stability classifications:

A = Extremely Unstable

B = Moderately Unstable

C = Slightly Unstable

D = Neutral, considered to be representative of average, long-term conditions

E = Slightly Stable

F = Moderately Stable

The combined conversion factors that describe particulate dioxin concentrations downwind from a source are as follows:

If $W < 10$ m/sec:

$$CF_{air} = CF_{particulate} = 1 \times 10^{-7} \left[\frac{(\sigma_y \sigma_z)_F}{(\sigma_y \sigma_z)_{x+F}}\right] Kg/m^3 \quad (8)$$

If $W \geq 10$ m/sec:

$$CF_{air} = CF_{particulate} = 1 \times 10^{-7} W^2 (W - 9) \left[\frac{(\sigma_y \sigma_z)_F}{(\sigma_y \sigma_z)_{x+F}}\right] Kg/m^3 \quad (7)$$

In all cases, chronic exposure is of greatest concern; therefore, wind velocity should reflect average annual wind speeds in the direction of interest rather than an instantaneous value. As a result, most sites will probably be evaluated at low wind speeds where vapor exposure will predominate. When mechanical disturbances are of consequence, however, representative values for $CF_{particulate}$ must be calculated and added to those for vapor to determine total exposure. More complex methods for rapid assessment of particulate emissions have been summarized by MRI (1984) and are recommended if time and resources allow.

3.4 Vapor Exposure

Dioxin has both a strong tendency to sorb on soils ($K_{ow} = 6.9 \times 10^6$, Mabey et al., 1981; $K_{ow} = 1.28 \times 10^7$, Perkaw et al., 1980) and a low vapor pressure, reported in the range 1×10^{-6} (Thibodeaux, 1983) to 1.5×10^{-9} mm Hg at 25°C (Freeman and Schroy, 1984). As a consequence, it has generally been assumed that particulate transport of adsorbed dioxin is the major route of movement. However, recent work suggests that dioxin in shallow soils (5 to 10 cm) may vaporize and disperse in the atmosphere (Freeman and Schroy,

1984; Thibodeaux, 1983), with ultimate photolytic decomposition (Nash and Beall, 1980).

In his study at the Vertac site, Thibodeaux (1983) calculated that vaporization of dioxin from the soil surface was the major route of dioxin loss from the site. Mass flux calculations, based on estimated values for pertinent environmental and chemical properties, predicted that vaporization losses from the site were much greater than losses from entrainment of soil particles.

Nash and Beall (1980) reported that dioxin volatilized from soil in microecosystem chambers and from field plots. Significant quantities of dioxin in the air from both experiments appeared to be dechlorinated. The researchers concluded that atmospheric photodegradation was occurring. The rates of both volatilization and degradation depended on the dioxin application formulation and the temperature of the systems.

Freeman and Schroy (1984) used vaporization processes to model dioxin movement in a soil column. However, the researchers suggest that photodegradation at the soil surface will dominate vaporization losses during daylight hours. Thus dioxin losses to the atmosphere should occur primarily at night, with rapid photodegradation the next morning.

No research has been conducted to date on atmospheric degradation of dioxin. Research results suggest that dioxin is lost from the soil, but the loss mechanism and environmental fate are only poorly understood. Although volatilization may be an important loss mechanism, potential photodegradation may reduce any environmental transport. If the degradation process is

occurring, as postulated by Nash and Beall (1980) or Freeman and Schroy (1984), the potential off-site exposure to vapor will be very low.

Due to the uncertainties in volatilization and photodegradation, it is not possible to derive a conversion factor for dioxin vapor air concentration. As more research is conducted, derivation of a conversion factor for vapor (CF_{vapor}) may be possible. This CF_{vapor} should be added to the CF_{air} presented in this report to estimate total atmospheric dioxin concentrations at potential exposure points.

4.0 SOIL CONCENTRATIONS

4.1 Introduction

Overland transport of contaminated soil via runoff is an important mechanism which contributes to the potential for human exposure to dioxin. Human activities such as farming, gardening, excavating, and recreation can result in dermal absorption of contaminants or ingestion of contaminated soil, particularly by children. In order to estimate potential exposures downflow from a source, an approximation of the soil loss from a source and the redeposition of contaminated soil away from a source must be calculated.

4.2 Dioxin Behavior in Soil

Soil at a source becomes contaminated by adsorption of dioxin. No data were found to quantitatively describe dioxin concentration versus soil characteristics, such as particle size or organic content. However, it is assumed that due to its high K_{ow}, dioxin will be adsorbed primarily on the organic fraction of the soil. It is this high affinity for organics in soil and low solubility in water that are believed to account for the vertical immobility of dioxin (Kearney, Woolson, and Ellington, 1972; Matsumura and Benezet, 1973). Because small particles have a higher surface-to-volume ratio than large particles, it is also assumed that the small particle-size fraction of the soil would have a higher contaminant concentration than a bulk soil sample. Walling (1983) summarizes the relationship of particle

size and organic content characteristics of eroded soil to those of the original soil in five test-plot studies. These data suggest that contaminants such as nutrients or pesticides may be enriched up to 1.5 times on clay-sized particles, and more than 2 times on the organic fraction. Lack of quantitative data concerning these phenomena, however, precludes incorporating them into the derivation of conversion factors. It has been assumed, therefore, that all transported soil has the same dioxin concentration as determined for bulk soil samples from the site.

4.3 Photodegradation

Photodegradation is another process which may affect the amount of dioxin available for transport from a site and the amount to which humans may be exposed. Ultraviolet wavelengths have been shown to be effective in photodegrading dioxins. Photolysis apparently removes one or more chlorine atoms from the dioxin molecule, thereby making it less toxic but not destroying the basic dioxin nucleus (Crosby et al., 1971).

Esposito et al., (1980) provide a comprehensive review of numerous photodegradation studies and the inconsistent results. Crosby et al., (1971) applied dioxin to several matrices. Although decomposition was rapid in alcohol solution, there was negligible loss from aqueous suspension and wet or dry soil after 96 hr of irradiation. However, the researchers suggested that in the natural environment, waxy leaf cuticles, surface slicks on water, and spray oils or solvents commonly incorporated in pesticide formulations may serve as the organic hydrogen donors necessary for photodegradation.

Dioxin applied to soil and exposed to artificial sunlight (sunlamp) for 96 hr showed no degradation, as reported by Crosby, Moilanen, and Wong

(1973). In other studies (Crosby and Wong, 1977), dioxin-contaminated Herbicide Orange was applied to plant leaves and soil and exposed to sunlight. After 6 hr of exposure, 0-30% of the dioxin remained on the plant leaves, with 30% remaining on soil which had received the lowest applied concentration (1.3 ng/cm^2). At the application rate of 10 mg/cm^2 approximately 90% of the dioxin remained after 6 hr of exposure. The researchers believed that surface soil particles shaded the underlying particles, thereby preventing photodecomposition at depth. It was concluded from the 1977 study that the three requirements for dioxin decomposition/dechlorination are: 1) dissolution in light-transmitting film; 2) the presence of an organic hydrogen donor, such as solvent or pesticide; and 3) ultraviolet light. All three conditions should be present in the application or accidental loss of materials commonly contaminated with dioxin (2,4,5-T, trichlorophenol, PCB road oils). Crosby and Wong conclude that dioxin is not stable as a contaminant in thin herbicide films exposed to outdoor light.

In response to the work by Crosby and co-workers, photodegradation was evaluated as a decontamination technique in Seveso, Italy (Liberti et al., 1978). Exposure of dioxin-contaminated soils to artificial ultraviolet light and natural sunlight in the presence of a hydrogen donor resulted in degradation at the surface and to a certain extent, degradation beneath the soil surface. The degradation rate in soil from natural sunlight would be affected by sunlight intensity, nature of the contaminated medium, temperature, and the amount of vegetative cover at a site.

Although photodegradation may have a significant effect on environmental dioxin concentrations, it was not included in deriving soil conversion factors. The amount of degradation appears to depend on site-specific factors (sunlight intensity, temperature, substrate, co-contaminants) that are not amenable to a generic approach. If photodegradation is occurring, the method presented in this report will overestimate the dioxin concentrations at the point of exposure.

4.4 Volatilization

As discussed in Chapter 3.0, volatilization from a site may be a significant loss mechanism. Nash and Beall (1980) and Thibodeaux (1983) report that volatilization may be a major pathway. Matsumura and Ward (1976) indicate that the water content of soil may mediate the evaporation rate. The effects of volatilization on concentrations at a site and at points of exposure have not been considered in deriving CF_{soil}, so overestimation of soil exposure concentrations may result when using this method.

4.5 Approach

Empirical and deterministic methods were evaluated for applicability in deriving conversion factors. The empirical approach involved collecting monitoring data from specific sites and trying to correlate observed distributions with characteristic site parameters. Because it was expected that few comprehensive sets of dioxin data would be available, monitoring data for other persistent contaminants such as polychlorinated biphenyls (PCBs), polybrominated biphenyls (PBBs), heavy metals, and radionuclides were also sought.

Only two sites were found where sampling had been conducted at and away from a site. Roberts, Cherry, and Schwartz (1982) studied the distribution and surface translocation of a serious PCB spill at a transformer manufacturing plant in Regina, Saskatchewan, Canada. The researchers found that particle transport in runoff from eroding areas was an important migration mechanism. However, PCB distributions were "extremely heterogeneous," with "no definable trends in concentrations."

Dioxin contamination in Seveso, Italy, was caused by wind-influenced atmospheric deposition from the 1976 explosion at the Givaudan-LaRouche ICMESA plant. Sampling was conducted within 110 hectares southeast of the site for over 3 years (DiDomenico et al., 1980). Dioxin concentrations at locations 100 m apart varied by as much as a factor of 100, and this highly irregular distribution changed very little during the three-year study period. Although the mechanism by which dioxin was initially distributed differs from that characteristic of uncontrolled disposal sites, the method of transport from the originally contaminated area is similar to the problem addressed in this report. Of particular interest are the following:

- Areas of high contamination showed little dioxin contamination reduction over three years.
- Slightly contaminated or uncontaminated areas downwind and within runoff routes showed no statistically-significant increase in dioxin concentrations over three years.

Assessment activities at dioxin-contaminated sites in Missouri and Arkansas did not include systematic sampling/analysis at the sites and at intervals away from the sites. Sampling was not conducted at known high

concentration sources scheduled for remedial action at the Vertac site (JRB, Inc., 1983). Sampling was conducted at numerous sites in Missouri, but only to locate areas of high concentration. Few samples were acquired from each site and no descriptions of site or pathway characteristics were provided (U.S. EPA, 1982a; U.S. EPA, 1982b).

Based on available site data, it was not possible to derive correlations between dioxin concentrations at sources and at points of exposure. The Regina and Seveso data provide qualitative indications that contaminant concentration distributions will probably be irregular and thus difficult to predict. Belli et al., (1983) report that the statistical analysis of data from regions of low contamination at Seveso was most strongly affected by the sensitivity and precision of analytical instrumentation.

Deterministic approaches involve mathematical modeling of the physical transport process from a site to a selected exposure point. Onishi, Whelan, and Skaggs (1982) present a review of overland soil and sediment transport models and divide them into three groups based on their degree of complexity and the extent to which they represent physical processes.

The simplest models require the least amount of site-specific data and use an empirical formula to estimate average soil losses from an area. A modified version of the Wischmeier and Smith (1978) USLE requires limited data on watershed characteristics.

The second group of models requires considerable amounts of detailed hydrologic, meteorologic, and site-specific physical characteristics to simulate soil erosion and transport. If the required data are available, these models are generally more accurate than the simplest, empirical models.

Both the first and second model groups account for chemical distributions via loading factors.

The final group of models simulates environmental chemical behavior, such as adsorption-desorption and decay, as well as runoff and erosion. In addition to the data required for Group 2 models, the most complex models require chemical characteristics and distributions on the land surface.

The last two groups of models are useful only to those who have access to a digital computer, and are therefore of no interest to those wishing to calculate simple conversion factors. In addition, these models require detailed site and chemical data that are generally not available without extensive field investigations.

The model used to derive conversion factors from average soil loss and deposition is consistent with the first group of models and does not require a computer or cumbersome amounts of site-specific data. Because chronic exposure is of primary interest, average loss and deposition are appropriate for assessing lifetime exposure, rather than for cyclic or acute events. The approach utilizes the USLE.

4.6 Average Soil Losses

Average annual soil losses from a contaminated site can be approximated by using the USLE, an empirical formula which was developed for agricultural land using data from numerous field test soil plots (Wischmeier and Smith, 1978). The equation input factors have been modified slightly for use in areas other than cropland. The USLE provides an approximation of sheet and rill erosion losses, in tons per acre per year, due to the interaction of six physical factors which can be expressed numerically as site characteristics.

Tables and maps are provided for use in selecting site-specific values for these factors.

The USLE defines loss as:

$$L = R \times K \times LS \times C \times P \qquad (28)$$

where L = computed average annual soil loss (tons/acre/yr)

R = rainfall erosion index (yr^{-1})

K = soil erodibility factor (tons/acre per unit of rainfall factor, R)

LS = topographic factor (dimensionless)

C = cover and management factor (dimensionless)

P = support practice factor (dimensionless)

Useful procedures for the estimation of USLE parameters for both agricultural and non-agricultural conditions can be found in Mills et al., (1982).

The average annual soil loss per unit area, L, represents an average annual value and is obtained by multiplying the rainfall erosion index (which provides estimated soil losses due to rainfall and runoff for a geographic area) by a series of ratios. These ratios represent the relationship of actual parameters to those observed in test soils and standardized agricultural plots.

The rainfall erosion index, R, expresses erosion potential for average annual rainfall at a location. A map of average R values for the U.S., based on over 30 yrs of measurements, is provided in Figure 6. Interpolation between contour lines is necessary for many areas of the country.

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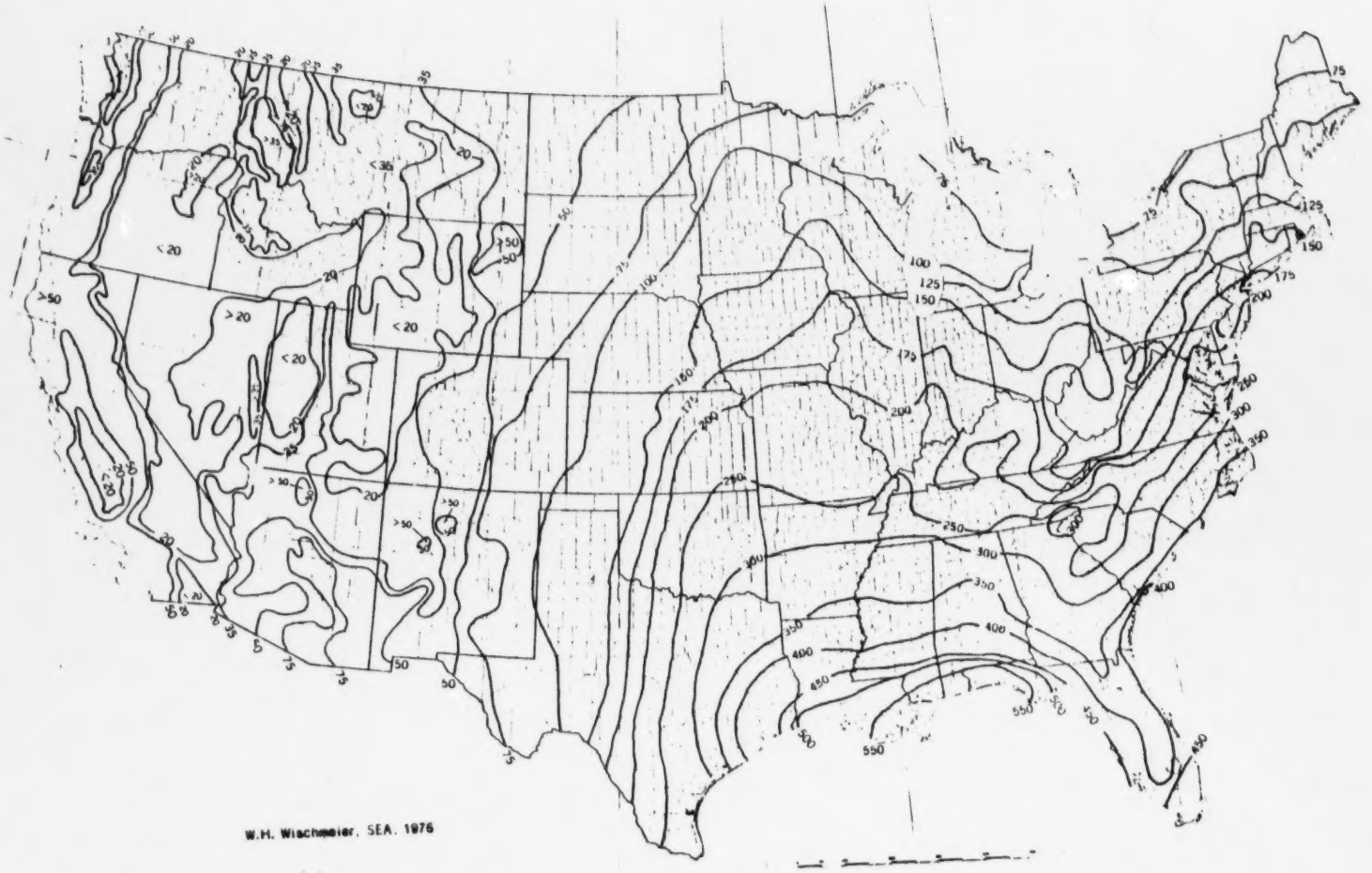

Figure 6. Average Annual Values of the Rainfall Erosion Index (yr^{-1}) (from Wischmeier and Smith, 1978)

Values for K, the soil erodibility factor, have been experimentally determined for a number of benchmark soils at erosion research stations in the U.S. Average values of K, based on a range of soil types, are provided in Tables 2 and 3. The soil erodibility for a particular site can be approximated by using the K value corresponding to the predominant soil type. Average values for basic soil types are provided in Table 2. Assuming soil organic content is known or can be estimated, more specific values for K are available in Table 3.

The topographic factor, LS, combines the effect of slope length and steepness. Values for the area under consideration can be determined using the average percent slope and slope length, measured in ft. A listing of LS values for slopes of varying gradients and lengths is provided in Table 4. Interpolation between listed values may be necessary.

The cover and management factor, C, is most significant for agricultural land where it is a function of vegetative cover, crop sequence, crop rotation, and tilling practices. Wischmeier and Smith (1978) provide guidelines for determining C values for construction sites, pasture, range, idle land, and forested areas. In order to simplify site characterization, two C values have been selected. A C value of 1.0 represents a worst-case scenario and should be used when vegetation is completely absent. Examples of this type of site would be horse arenas, unpaved roads, and unvegetated landfills. A C value of 0.5 should be used for any other type of site. Because 0.5 represents a high value for permanent pasture, range, wooded, and idle land, a worst-case scenario for vegetated land has been assumed. For wooded areas with highly erodible soil and no surface vegetative cover, the C

Table 2. Average Values for the Soil Erodibility Factor, K, for Soils on Erosion Research Stations (After Wischmeier and Smith, 1978)

Average Soil Type	K Value (tons/acre)
Silt Loam Loam Sandy Clay Loam	0.4
Silty Clay Loam Clay Clay Loam	0.3
Fine Sandy Loam	0.2
Loamy Sand Flaggy Silt Loam	0.1
Gravelly Loam	<0.1

Table 3. General Magnitude of the Soil/Erodibility Factor, K*, when Organic Content Data are Available (Carsel et al., 1984)

Texture Class	Organic Matter Content <0.5%	2%	4%
Sand	0.05	0.03	0.02
Fine Sand	.16	.14	.10
Very Fine Sand	.42	.36	.28
Loamy Sand	.12	.10	.08
Loamy Fine Sand	.24	.20	.16
Loamy Very Fine Sand	.44	.38	.30
Sandy Loam	.27	.24	.19
Fine Sandy Loam	.35	.30	.24
Very Fine Sandy Loam	.47	.41	.33
Loam	.38	.34	.29
Silt Loam	.48	.42	.33
Silt	.60	.52	.42
Sandy Clay Loam	.27	.25	.21
Clay Loam	.28	.25	.21
Silty Clay Loam	.37	.32	.26
Sandy Clay	.14	.13	.12
Silty Clay	.25	.23	.19
Clay		0.13-0.29	

*The values shown are estimated averages of broad ranges of specific-soil values. When a texture is near the borderline of two texture classes, use the average of the two K values. For specific soils, Soil Conservation Service K-value tables will provide much greater accuracy.

Table 4. Values of the Topographic Factor, LS, for Specific Combinations of Slope Length and Steepness (From Wischmeier and Smith, 1978)

Percent slope	Slope length (feet)											
	25	50	75	100	150	200	300	400	500	600	800	1,000
0.2	0.060	0.069	0.075	0.080	0.086	0.092	0.099	0.105	0.110	0.114	0.121	0.126
0.5	.073	.083	.090	.096	.104	.110	.119	.126	.132	.137	.145	.152
0.8	.086	.098	.107	.113	.123	.130	.141	.149	.156	.162	.171	.179
2	.133	.163	.185	.201	.227	.248	.280	.305	.326	.344	.376	.402
3	.190	.233	.264	.287	.325	.354	.400	.437	.466	.492	.536	.573
4	.230	.303	.357	.400	.471	.528	.621	.697	.762	.820	.920	1.01
5	.268	.379	.464	.536	.656	.758	.928	1.07	1.20	1.31	1.52	1.69
6	.336	.476	.583	.673	.824	.952	1.17	1.35	1.50	1.65	1.90	2.13
8	496	.701	859	.992	1.21	1.41	1.72	1.98	2.22	2.43	2.81	3.14
10	.685	968	1.19	1.37	1.68	1.94	2.37	2.74	3.06	3.36	3.87	4.33
12	.903	1.28	1.56	1.80	2.21	2.55	3.13	3.61	4.04	4.42	5.11	5.71
14	1.15	1.62	1.99	2.30	2.81	3.25	3.98	4.59	5.13	5.62	6.49	7.26
16	1.42	2.01	2.46	2.84	3.48	4.01	4.92	5.68	6.35	6.95	8.03	8.98
18	1.72	2.43	2.97	3.43	4.21	3.86	5.95	6.87	7.68	8.41	9.71	10.9
20	2.04	2.88	3.53	4.08	5.00	5.77	7.07	8.16	9.12	10.0	11.5	12.9

[1] $LS = (\lambda/72.6)^m (65.41 \sin^2\theta + 4.56 \sin\theta + 0.065)$ where λ = slope length in feet; $m = 0.2$ for gradients < 1 percent, 0.3 for 1 to 3 percent slopes, 0.4 for 3.5 to 4.5 percent slopes, 0.5 for 5 percent slopes and steeper; and θ = angle of slope. (For other combinations of length and gradient, interpolate between adjacent values or see fig. 4.)

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value would approach 1.0. Estimation of soil losses using the C values discussed here would probably result in higher rates than would be actually observed in the environment, but are deemed to be acceptable approximations.

The support practice factor, P, is also dependent on agricultural techniques and is a function of such practices as contouring and terracing. Because there is no counterpart to P on natural land or construction sites, the value of P has been set at 1.

Users of the USLE must be aware of its limitations (Wischmeier, 1976; Walters, 1983). Soil losses from a source or area can be determined by using the USLE; however, the USLE provides only an estimate of the amount of soil eroded from a specific area and does not indicate the amount of sediment actually delivered to streams. The sediment yield is the total amount of soil loss from the area less the amount of deposition which occurs.

R values obtained from Figure 6 (Wischmeier and Smith, 1978) are applicable only for long-term erosion averages. Values for K, the soil-erodibility factor (Tables 2 and 3), are averages for soil types, but the actual amount of soil loss for any soil type can vary widely as a function of antecedent soil moisture conditions. The amount of runoff will be significantly different for saturated and unsaturated soils.

The USLE was developed primarily from data obtained east of the Rocky Mountains, so its applicability to the arid western states may be somewhat limited. Use of the USLE may result in significant errors due to the predominance of high intensity, short duration rainfall in the West, and the greater effect of other physical conditions such as wind, humidity, and heat.

Because the USLE was developed using data from small field plots, it predicts sediment yields of particles of 1-mm diameter (coarse sand) and finer sediments. The USLE is not applicable to coarser sand, gravel, and larger particles.

4.7 Soil Deposition

A Modified Universal Soil Loss Equation (MUSLE) (Simons, Li, and Associates, 1982; Walters, 1983) has been developed for determining single-storm event sediment yields from drainage basins. The substitution of a runoff factor for the rainfall factor, R, in the USLE makes the MUSLE better suited for use in areas west of the Rocky Mountains. Use of the MUSLE, however, requires calculating site-specific coefficients, which preclude its general use for determining sediment yields. The MUSLE has been further modified for computing annual sediment yield, but this calculation is also site-specific based on weighted storm yields for selected return periods.

To determine sediment yield to a stream, the sediment delivery ratio is used (Piest and Miller, 1975):

$$D = Y/L \tag{29}$$

where D = sediment delivery ratio, the change per unit area of sediment delivery downstream (dimensionless)

Y = sediment yield at measuring point (tons/acre/yr)

L = total amount of sediment eroded from drainage area upstream of measuring point, estimated using the USLE (tons/acre/yr).

To date there have been no comprehensive studies defining sediment delivery relationships for the U.S. on a regional basis. It is impossible to define relationships which would hold true for all geographic areas. Walling (1983) states that the processes of sediment delivery are very complex and are dependent on a variety of factors, including topography of the source area, stream channel characteristics, drainage patterns, vegetative cover, land use, soil properties, and the distribution of sediment sources. Interrelationships between these factors are difficult to define, and errors can be introduced because measured sediment yields are compared to total erosion from a source estimated with a generalized soil loss equation.

As summarized by Walling (1983), there is evidence that only a small percentage of the drainage basin area provides storm runoff in humid regions, and the actual runoff area for the same delivery location varies in extent and location depending on antecedent moisture conditions. This evidence suggests that the sediment delivery ratio is dependent on only the characteristics of that portion of the drainage basin which produces storm runoff and would change with time as the area changes.

Some relationships have been characterized sufficiently to show general trends between the size of the drainage basin and the sediment delivery ratio. Piest and Miller (1975) present a summary of this relationship in Figure 7. Walling (1983) provides a curve showing the relationship between the sediment delivery ratio and drainage basin area for the central and eastern U.S., as developed by the U.S. Department of Agriculture Soil Conservation Service. He also gives a summary of 10 relationships from selected drainage basins in the U.S. and other countries as well as a summary

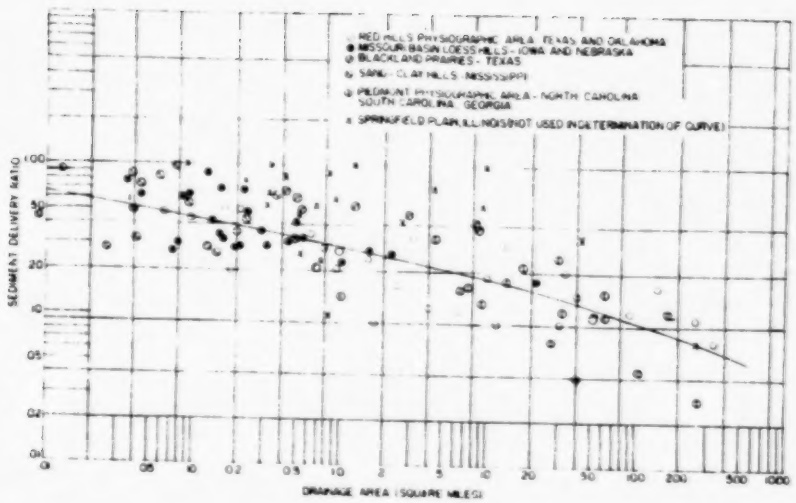

Figure 7. Generalized Relationship Between Size of Drainage Basin and Sediment Delivery Ratio (from Piest and Miller, 1975)

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of 13 sediment yield/drainage basin area relationships which are directly analogous to the delivery ratio/drainage basin area relationship (Piest and Miller, 1975; Walling, 1983). Some of these relationships show marked similarities to the relationship in Figure 7.

The relationship shown in Figure 7 is generally applicable to the central and eastern U.S. No comparable data could be found for the western U.S. The sediment delivery ratios shown in Figure 7 vary widely for a given drainage basin area. The values for basins often vary by a factor of 2, and sometimes by an order of magnitude.

It is assumed that sediment delivery ratio is inversely proportional to drainage basin size because of greater redeposition that will occur as sediment travels over greater distances before reaching the point of interest. It can be concluded by analogy that redeposition of contaminated soil will become smaller as locations of interest are more distant from the source site.

The concentration of dioxin in soil at a point of interest, x, is:

$$C_{soil} = \frac{\text{mass of dioxin delivered to x}}{\text{mass of soil delivered to x}} \tag{30}$$

$$= \frac{(\text{mass of dioxin lost from source})(\text{fraction delivered to x})}{\text{mass of soil delivered to x}} \tag{31}$$

The fraction delivered from the source to point x is the sediment delivery ratio for an assumed watershed which begins at the source and encompasses the natural drainage area between the source and the point of interest. The mass of soil lost from the source can be estimated using the

USLE and the source area. The total mass of soil delivered to point x can be estimated using the USLE, the watershed area, and the sediment delivery ratio for the entire watershed area. Frequently, USLE estimates will be based on the average slope and slope length for the watershed. The expression for soil concentration, thus, becomes:

$$C_{soil} = \frac{C_{source} \times L_{source} \times A_{source} \times D_{source}}{L_{basin} \times A_{basin} \times D_{basin}} \qquad (32)$$

where C_{source} = dioxin concentration in soil at the source (ug/Kg)

L_{source} = estimated soil loss from the USLE for the source (tons/acre/yr)

A_{source} = source area (acres)

D_{source} = sediment delivery ratio for the area between the source and the point of interest (dimensionless)

L_{basin} = estimated soil loss from the USLE for the watershed upstream of the point of interest (tons/acre/yr)

A_{basin} = watershed area upstream of the point of interest (acres)

D_{basin} = sediment delivery ratio for the watershed area upstream of the point of interest (dimensionless)

From Figure 7, it appears that the sediment delivery ratio for a specific basin area can range from about 0.1 or 0.2 to 1.0, depending on factors other than basin area. On the other hand, the delivery ratio as a function of basin size only varies over about the same range. Therefore, in general, the sediment delivery ratio for the source and watershed basin areas are not considered to be significantly different. If sediment delivery

ratios at the point of interest are not known, C_{soil} can nevertheless be estimated using a simplified form of equation 45:

$$C_{soil} = C_{source}\left[\frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}}\right] \quad (9)$$

In areas where the downgradient points of exposure lie in the downwind direction from the site, it will be necessary to consider the effect of atmospheric concentrations, as well as soil concentrations. Atmospheric concentrations and the corresponding conversion factor were discussed in Chapter 3.0.

4.8 Wind Deposition

The constructs for calculating C_{air} presented in Chapter 3.0 incorporate the assumption that the soil concentration of dioxin (C_{soil}) is known at the location of interest. If observed data are not available, a means of calculating downwind soil concentrations arising from deposition during previous storm events will be necessary.

For the purposes of this report, it is assumed that dioxin-contaminated soils in downwind areas arise from windblown particulates. Although some vapor transport may occur, this process was neglected due to the uncertainties discussed in Section 3.4. If the transported particles of interest are in the range $\leq$ 20 micron, they are subject to dilution as predicted by a Gaussian distribution for the plume. Because respirable particles are $\leq$ 10 micron, this assumption incorporates all particulates of interest. Larger particles containing dioxin will settle more rapidly and therefore reduce atmospheric dioxin levels. Hence, this assumption may lead

to overprediction. In this case, atmospheric concentrations, C_{air}, at point x can be calculated using the following formula (Turner, 1970):

$$C_{air(x,o,o)} = \frac{Q_z}{\pi\sigma_y\sigma_z W} \tag{24}$$

where Q_z = emission rate (gm/sec)

$\sigma_y\sigma_z$ = Product of the Gaussian dispersion coefficients (m^2), from Figure 4

W = wind speed (m/sec)

Therefore, the ratio of concentrations at two points can be determined by the ratio:

$$\frac{C_{air1}}{C_{air2}} = \frac{(\sigma_y\sigma_z)_2}{(\sigma_y\sigma_z)_1} \tag{25}$$

If the phenomena that relate atmospheric levels to soil concentrations at a given location are essentially the same for all points in the downwind direction (i.e., $CF_{air} = C_{air}/C_{soil}$), it holds that

$$\frac{C_{soil1}}{C_{soil2}} = \frac{(\sigma_y\sigma_z)_2}{(\sigma_y\sigma_z)_1} \tag{33}$$

Values for $\sigma_y\sigma_z$ downwind from a source are given in Figure 4 for the six stability classifications. Assuming that most particulate transport arises from major storm events, the $\sigma_y\sigma_z$ values for Stability Class A are most appropriate for predicting downwind soil concentrations. Therefore, downwind soil dioxin levels at distance x can be calculated using the following relationship:

$$CF_{soil} = \frac{C_{soil}}{C_{source}} = \frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \quad (10)$$

where F = fetch or downwind dimension of the source (m)

x = distance from source boundary to the point of interest (m)

$\sigma_x\sigma_z$ = product of the Gaussian dispersion coefficients (m^2), from Figure 4.

When runoff patterns and prevailing wind direction are coincident, soil concentrations should be based on the summation of the two contributions, as follows:

$$CF_{soil} = \frac{C_{soil}}{C_{source}} = \frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} + \frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}} \quad (11)$$

This surface concentration value relates to the top centimeter of soil and should be used for subsequent calculation of downwind dioxin-particulate levels in the atmosphere, assuming that no soil mixing occurs prior to resuspension.

5.0 SEDIMENT CONCENTRATIONS

5.1 Introduction

Human health risks may arise when individuals are exposed to dioxin through the consumption of dioxin-contaminated fish. Fish accumulate dioxin from an aquatic environment in two ways (Isensee and Jones, 1975). Bottom feeding species, in particular, may ingest contaminated sediment along with their food. Any fish species can accumulate dioxin directly from the water. In these cases, dioxin is desorbed from contaminated sediments or absorbed from stream water. To quantify human exposure, it is necessary to develop a means of approximating the concentrations of dioxin in stream sediments in the vicinity of a contaminated site. The conversion factor described in this report can be used in the algorithm developed by Schaum (1984) to estimate the bioconcentration of dioxin in various fish species and the subsequent human exposure.

5.2 Dioxin Behavior in Water

Some of the processes which can affect dioxin when it is exposed to air (as discussed in Chapters 3.0 and 4.0) are expected to have minimal effects on dioxin in an aquatic environment. Crosby et al. (1971) report that volatilization does not appear to be of major importance in water. Other researchers report that evaporation from or with water may be a major cause of the disappearance of dioxin in a model aquatic ecosystem (Ward and

Matsumura, 1978), but the experimental results and their application to the natural environment are as yet inconclusive. Photodegradation is thought to occur so slowly as to be negligible in water (Crosby, Moilanen, and Wong, 1973; Crosby et al., 1971; Isensee and Jones, 1975; Matsumura and Ward, 1976). Some evidence of microbial degradation under experimental conditions has been documented; however, dioxin in water is generally thought to resist microbial effects (Ward and Matsumura, 1978; Matsumura and Ward, 1976). The half-life of dioxin was found to be on the order of 600 days in a model aquatic ecosystem (Ward and Matsumura, 1978).

When present at very low concentrations on sediment, dioxin is generally not expected to desorb due to its low solubility. At concentrations as low as 0.1 ppb, however, dioxin can desorb. Isensee and Jones (1975) report that under experimental conditions the concentrations of dioxin in water and sediment reached equilibrium in 4 to 15 days. The temporal variation was attributed to the difference in adsorption capacities of the two soils used.

5.3 Sediment Transport in Streams

Contaminants can be transported in streams by three processes: 1) as dissolved compounds in stream water, 2) as compounds adsorbed onto sediments and transported as suspended load, and 3) as compounds adsorbed onto sediments and transported as bed load. The low solubility and high affinity of dioxin for soils, particularly soils high in organic content (Isensee and Jones, 1975; Kearney, Woolson, and Ellington, 1972), suggest that dioxin would be transported primarily in an adsorbed phase on stream sediments. No data could be found on preferential adsorption of dioxin to any particular sediment particle size, so it was assumed that dioxin would be adsorbed

equally on all available particle size fractions. Pritchard (1984) reports that for polynuclear aromatic hydrocarbons, partition coefficients and natural transport of sediments adequately accounted for the observed distribution of the contaminants in an aquatic environment.

The source of stream sediment contamination in the vicinity of a contaminated site is surface soil on which dioxin is adsorbed. Such contaminated soil may reach a stream by direct stream erosion of the soil or, more commonly, by overland sediment transport. The latter process is known as sheet and rill erosion and occurs during runoff of precipitation. This type of sediment transport was discussed in Chapter 4.0. Of the soil eroded and transported by overland processes, some can be expected to reach both major and minor streams within a drainage system.

Finer soil particles, such as clay, silt, and fine sand, that reach the stream are usually transported as part of the suspended sediment load. Coarser particles, such as coarse sand and gravel, are usually transported as bed sediment load or are deposited in the stream bed.

Most streams normally flow at less than their capacity. This normal flow is called the mean annual discharge and corresponds to a water depth of only about one-third of the stream capacity or bank-full depth. Mean annual discharge is equaled or exceeded on an average of 25% of the time (Leopold, Wolman, and Miller, 1964).

For streams in general, the amount of suspended sediment varies logarithmically with respect to stream discharge. As discharge increases, streams can also transport larger sediments. Correspondingly, the majority of sediment transport in a given stream occurs during high flow conditions

and seasonal floods, rather than during very low or even normal flow conditions. During the higher flow events, particles previously transported as bed sediment load and sediments deposited in the stream bed may be added to the suspended sediment load. Extremely high flow events may transport very large quantities of sediments, but are so infrequent and of such short duration that their effect on the average sediment discharge is minimal (Longwell, Flint, and Sanders, 1969).

Studies of PCB concentrations and transport in the Hudson River in New York State (Turk, 1980) show a constant transport rate of PCBs during moderate non-flood discharges and increased transport during floods. PCB concentrations were found to increase as discharge fell below an intermediate value. At low discharge, resuspension of bottom material was minimal, but less dilution of contaminants occurred. At higher discharges, increased concentrations were due to resuspension of contaminated bottom sediments. During intermediate discharges, PCB concentrations were found to be a function of both sediment resuspension and dilution. However, the net effect of these opposing influences produces concentrations less than those achieved during either low or high stream discharges.

Deterministic and empirical approaches were examined for their usefulness in deriving exposure factors for stream transported sediments. In general, deterministic approaches involve the use of numerical models to approximate natural physical processes. Onishi, Whelan, and Skaggs (1982) provide a comprehensive review of a number of models which could be used to simulate sediment transport by streams. These models are divided into three groups, as described in Chapter 4.0. The least complex models require the

least amount of site-specific data. Stream transport models in this group involve dilution of contaminant concentrations with increases in stream discharge and distance downstream from the source. No allowance is made in these models for adsorption/desorption factors.

The two remaining groups of models require much more site-specific data, including detailed stream channel and flow characteristics, as well as adsorption/desorption and contaminant degradation factors. These models are best applied when such specific information is available and are unsuited for characterization of transport processes operating over a wide range of geographic areas where site generalization must be made.

The unsuitability of the last two groups of models for deriving simple conversion factors stems from the extensive, site-specific data requirements and the complexity of the computations. However, even the first group of simple models requires flow rates and sediment size distributions for each stream-transport scenario being considered. The sediment transport rate is derived using stream-specific characteristics and empirically derived constants that must be estimated for each sediment size range. Thus, none of the models described by Onishi, Whelan, and Skaggs (1982) are applicable for estimating non-stream-specific sediment transport.

Procedures utilizing USLE losses and sediment delivery ratios to estimate the sediment yield to streams from a source area have been discussed in Chapter 4.0. In order to determine the concentration of dioxin on sediments delivered to a stream, or at any point downstream of the source area, it is necessary to estimate what fraction of the total sediments at that point were derived from the source area. Consequently, the size of the

source area and the concentration of dioxin on the soil at the source must be known. The size of the watershed must also be known or estimated from topographic maps. Mechanisms of sediment supply, transport, and deposition within the drainage basin are assumed to be in equilibrium.

The sediment delivery ratio for a given drainage basin decreases with distance downstream of a contaminated source ; therefore, with distance downstream, a decreasing portion of the total sediment yield reaches the streams in the drainage area. The concentration of dioxin in stream sediments at the point of exposure is a function of the downstream decrease in sediment yield, due to deposition of contaminated sediments along the path of sediment transport between the source area and the point of exposure. The relationship between sediment delivery ratio and drainage area can and should be regionalized when applied to a given site due to the effect of features such as dams. The required data for regionalization are available for some watersheds.

Because the processes described in Section 4.7 are the same as those affecting sediment transport, the dioxin concentration relationship for sediments in a drainage system is:

$$CF_{sediment} = \frac{C_{sediment}}{C_{source}} = \frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} \qquad (12)$$

where C_{source} = dioxin concentration in soil at the source (ug/Kg)

$C_{sediment}$ = dioxin concentration in sediment at point of interest (ug/Kg)

L_{source} = estimated soil loss from the USLE for the source (tons/acre/yr)

A_{source} = source area (acres)

L_{basin} = estimated soil loss from the USLE for the watershed upstream of the point of interest (tons/acres/yr)

A_{basin} = watershed area upstream of the point of interest (acres)

6.0 MISSOURI BEEF DISTRIBUTION PATTERNS

Human exposure to dioxin through the consumption of beef products may result if the livestock ingested and accumulated dioxin as a result of contact with contaminated soils. Although dioxin exposure may occur from consumption of dairy products from contaminated cattle, only meat consumption is considered in this discussion. In areas where beef consumption involves locally grown and fed cattle, this pathway can be additive to those stemming from consumption of local fish, inhalation of dusts and vapor, and contact with (and/or ingestion of) soil. In order to quantify this potential pathway, an understanding of the pattern of beef production and meat processing in the area of interest is necessary. The area of interest for this report is the State of Missouri.

The beef industry in Missouri focuses largely on cow-calf production, i.e., grazing herds which are utilized to produce calves to a point of weaning. Backgrounding (preparing calves for feedlots) occurs in Missouri to a lesser extent, and feeding comprises only a small segment of the state beef industry. The small feeding segment is due, in part, to the fact that Missouri is a grain-deficit state and does not produce a sufficient excess of grain to economically support feeding operations. As a consequence, a large percentage of the Missouri calf crop is shipped to Nebraska and Kansas for backgrounding and feeding until it reaches marketable size.

The Missouri Crop Reporting Service reported that the January, 1984 herd consisted of 2,376,000 head of cows (Sestak, 1984). Approximately 25% of the herd is turned over through replacement, i.e., heifer calves are held back to replace death losses, dry and barren cows, or older animals which are slaughtered for low grade meat. The low grade meat is later distributed nationally for hamburger and could be shipped anywhere in the United States. Of the 25% of the herd turned over through replacement, roughly 60%, or 15% of the total herd, represent cows sent to slaughter for a variety of reasons. Total national input to this pool averages 7,000,000 head annually (USDA, 1984), so, the 594,000 replacement figure in Missouri constitutes 5% of the national inventory of cows destined for slaughter [(0.60)(594,000) ÷ 7,000,000].

At the replacement rate of 25%, 1,782,000 head of Missouri cows are available for transport to feeding. Of that number, roughly 150,000 are fed in state for commercial slaughter (Sestak, 1984). A second group of cows is held for home slaughter. Because there are 107,000 cattle ranches and beef-raising farms in Missouri (Grimes, 1984), with an estimated household size of 3.8 people, and assuming an average annual beef consumption level 77.4 lbs/capita (Berglund, 1984), home slaughter could account for 107,000 x 3.8 x 77.4 = 31,470,840 lbs/yr. This estimate is conservative because not all farms slaughter their own beef for personal consumption. If the average yield per head for home feeding is 550 lbs, 31,470,840 lbs/yr equate to 57,219 head. Hence, roughly 200,000 head of cattle are raised, fed, slaughtered, and consumed in Missouri each year. The remaining 1,582,000 head of calves are shipped out of state for feeding, the bulk of which are

sent to Nebraska and Kansas. In 1983, Nebraska marketed 4,580,000 head of fed cattle and Kansas marketed 3,410,000 head, for a total of 7,990,000 head (Gustafson, 1984). If it is assumed that the 1,582,000 head of Missouri-raised calves are uniformly mixed into this pool, fed, slaughtered, and distributed throughout the area as retail beef, approximately 20% of the beef imported into Missouri would have been calved in Missouri. This estimate is conservative because some beef may also be imported from Illinois and other neighboring states.

The fraction of cattle that leave Missouri and then return as wholesale and retail beef will have been subjected to "clean" feed during their confinement. The feed period often lasts as long as 6 months, or 24 weeks. Agricultural researchers have determined that the half-life of dioxin in beef is 16.5 $\pm$ 1.4 weeks (Jensen et al., 1981). Therefore, cattle fed out of state will have had a period of up to 1.5 half-lives to eliminate dioxin from their bodies. This will lead to an overall reduction of

$$C = C_o e^{\frac{-(\ln 2)t}{t_{\frac{1}{2}}}} \qquad (34)$$

$$C = C_o e^{\frac{-(\ln 2)24}{16}}$$

$$C = C_o e^{-1.04}$$

$$C/C_o = 0.35$$

where C = dioxin concentration at time of consumption

C_o = initial dioxin concentration.

Hence, the beef produced from those Missouri-raised calves that were sent to out-of-state feed operations and then returned to the state will carry 35% of their original dioxin levels. At the same time, the animals will have doubled in size; thus, 35% of the original mass of dioxin in these animals will be distributed over twice the total volume of beef, resulting in a dioxin concentration equal to 18% of the original contamination level C_o in the beef (i.e., $C/C_o = 0.18$).

Assuming an average annual consumption of 77.4 lbs/capita, the 5,000,000 people residing in Missouri will consume 390,000,000 lbs of beef each year, or the equivalent of 710,000 head. As noted above, 50,000 head will have been home slaughtered and 150,000 head will have been raised and fed in Missouri. Therefore, 510,000 head, or 72% of all beef consumed in Missouri (77% of the beef consumed by non-cattle-raising inhabitants) will have been imported from adjoining states.

Actually, none of the Missouri herd is known to be contaminated. However, if contamination were detected, a factor of H/H_o, where H is the size of the contaminated Missouri herd and H_o is the total Missouri herd size of 2,376,000, could be used to calculate potential market dilution effects. H should be estimated on the basis of animals on contaminated pasture. If the actual herd size is not known, it can be estimated based on acres of contaminated pasture and cattle density for the state (i.e., cows/acre pasture normal use).

Based on the considerations discussed previously, the following conclusions can be drawn with respect to dioxin exposure in Missouri from the consumption of beef:

- No natural degradation or market dilution will occur for the home-slaughter situation. Thus, any people consuming home-slaughtered beef where contamination is found would be in a high risk population with respect to beef consumption as a dioxin-exposure pathway.
- The remaining Missouri inhabitants who purchase wholesale or retail beef will consume beef consisting of 23% raised and fed in Missouri; 15% calved in Missouri but fed out of state, and 62% calved, raised, and fed out of state (Figure 8).

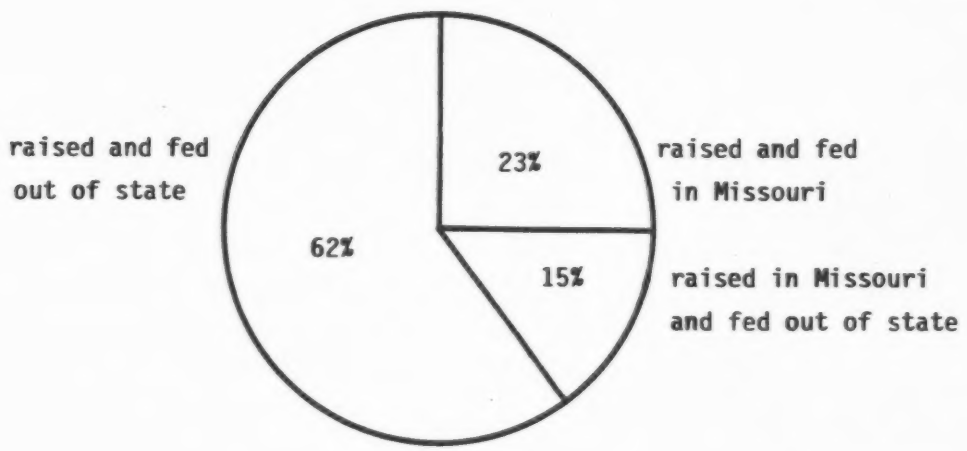

Figure 8. Origin of Beef Consumed in Missouri

- On the average, dioxin levels (C) in beef consumed by the non-cattle-raising inhabitants will be:

$$C = \left[(0.23)C_o + (0.15)(0.18)C_o + (0.62)(0)C_o\right] H/H_o \qquad (35)$$

$$= (0.23 + 0.027 + 0)(C_o)(H/H_o)$$

$$= 0.26(C_o)(H/H_o)$$

where C_o = predicted level of dioxin for beef raised entirely in a dioxin-contaminated area

H = size of the contaminated herd

H_o = total size of the state herd.

This relationship assumes that all out-of-state cattle are dioxin free.

The total herd size (H_o) in January, 1984 was 2,376,000, therefore:

$$C = (0.26)(C_o)(H/2,376,000)$$

$$= 1.1 \times 10^{-7} C_o H$$

- Approximately 5% of the U.S. slaughter cow inventory comes from Missouri. Thus, subsequent meat products such as hamburger may contain dioxin contamination at

$$C = 0.05 (H/H_o)C_o \qquad (36)$$

and for January, 1984 data

$$C = 2.0 \times 10^{-8} HC_o$$

It should be noted that if an entire beef or half a beef is purchased, or if a large amount of retail cuts are purchased at a single time, all of

the meat will be contaminated or contamination free depending on the source. When viewed in this manner, the market dilution concept appears inaccurate. However, when chronic exposure is considered, the market dilution concept is analogous to the purchase of a small percentage of dioxin-contaminated beef within a larger volume of total beef purchased.

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APPENDIX A

EXAMPLE SITE 1

A.1 SITE DESCRIPTION

Consider the case of a property where dioxin-contaminated soil was used as fill. The filled area (the source) is approximately 100 ft (30.5 m) long and 100 ft (30.5 m) wide (0.23 acre). Sampling indicated that the average dioxin concentration in the source was 150 ug/Kg. The source is currently without vegetative cover.

The property is located in a valley through which a creek flows (Figure A-1). The source is about 50 ft (15 m) from the creek. The slope of the property is 1%. Soil type in the area is primarily clay. Average annual wind speed is 5 mph (2.24 m/sec), with the predominant wind direction down the valley.

A.2 PROBLEM

Sampling was conducted only in the source area where human health risks were considered to be highest. However, potential exposure to dioxin is also of interest for areas downwind, downslope, and downstream of the site. Concentrations of dioxin at such points of exposure can be predicted using the appropriate conversion factors for the various modes of environmental transport.

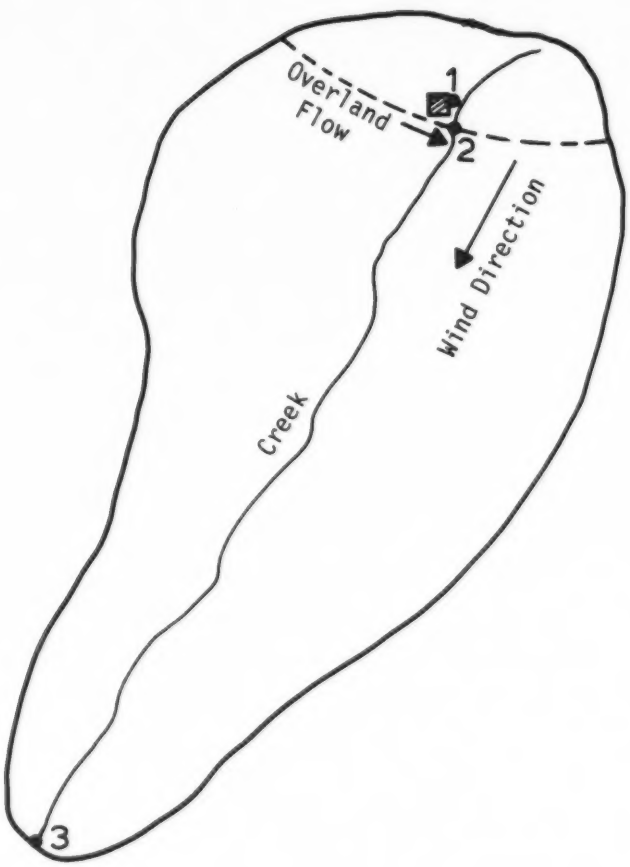

Figure A-1. Example Site 1

In this example, there is concern for dioxin concentrations at the following locations:

1) atmospheric concentrations at the source;
2) soil and atmospheric concentrations near the creek; and
3) sediment concentrations in the creek 1,000 ft (300 m) down the creek from the source.

The drainage basin area at Point 2 is 0.05 sq mi (32 acres). The average slope in the area is 1%, with an average slope length of 200 ft (61 m).

The drainage basin area at Point 3 is 0.7 sq mi (448 acres), with an average slope of 2% and an average slope length of 500 ft.

A.3 ATMOSPHERIC CONCENTRATION AT THE SOURCE

Because the average wind speed is less than 10 m/sec, Equation 21 is used to calculate air concentrations.

$$CF_{air} = \frac{C_{air}}{C_{soil}} = 1 \times 10^{-7} \text{ Kg/m}^3 \qquad (21)$$

$$\begin{aligned} C_{air} &= (1 \times 10^{-7} \text{ Kg/m}^3)\ C_{soil} \text{ ug/Kg} \\ &= (1 \times 10^{-7} \text{ Kg/m}^3)(150 \text{ ug/Kg}) \\ &= 0.000015 \text{ ug/m}^3 \\ &= 15 \text{ pg/m}^3 \end{aligned}$$

A.4 SOIL CONCENTRATION NEAR THE CREEK

The volume of soil lost from the site source is:

$$L = R \times K \times LS \times C \times P \text{ tons/acre/yr} = 12 \text{ tons/acre/yr}$$

R = 215 yr^{-1} (interpolated from Figure 6)

K = 0.3 tons/acre (from Table 2)

LS = 0.186 (interpolated from Table 4)

C = 1

P = 1

Because the source area is 0.23 acre, the mass of soil from the site is 2.76 ton/yr (2,500 Kg/yr). At a soil dioxin concentration of 150 ug/Kg, 376 mg of dioxin are transported from the source annually.

The volume of soil lost from the drainage area above Point 1 is:

$$L = R \times K \times LS \times C \times P = 7.35 \text{ tons/acre/yr}$$

where R = 215 yr^{-1}

K = 0.3 tons/acre

LS = 0.228

C = 0.5

P = 1

With a drainage basin area of 32 acres, the mass of soil lost from the basin is 235 tons/yr (213,000 Kg/yr).

The dioxin concentration in soil at Point 2 is:

$$C_{soil} = C_{source}\left[\frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}}\right] \qquad (9)$$

$$= (150 \text{ ug/Kg})\left[\frac{12 \text{ tons/acre/yr} \times 0.23 \text{ acre}}{7.35 \text{ tons/acre/yr} \times 32 \text{ acre}}\right]$$

$$= 150 \text{ ug/Kg } \frac{2.76}{235}$$

$$= 1.8 \text{ ug/Kg}$$

Because the predominant wind direction is down the valley (at right angles to overland flow direction), atmospheric deposition need not be considered when calculating soil concentrations at Point 2.

A.5 ATMOSPHERIC CONCENTRATION NEAR THE CREEK

Because the predominant wind direction is down the valley (at right angles to overland flow direction), dioxin in the soil at Point 2 will be the primary source of atmospheric contaminants. At an average wind velocity of 2.24 m/sec, atmospheric concentrations can be calculated using Equation 21, as follows:

$$C_{air} = (1 \times 10^{-7} \text{ Kg/m}^3) C_{soil} \text{ ug/Kg} \qquad (21)$$

$$= (1 \times 10^{-7} \text{ Kg/m}^3)(1.8 \text{ ug/Kg})$$

$$= 0.00000018 \text{ ug/m}^3$$

$$= 0.18 \text{ pg/m}^3$$

A.6 SEDIMENT CONCENTRATION DOWNSTREAM

Soil losses from the source are the same as those calculated in Section A.4. Soil losses from the basin upstream of Point 3 are:

$$L = R \times K \times LS \times C \times P = 10.5 \text{ tons/acre/yr}$$

where $R = 215 \text{ yr}^{-1}$

$K = 0.3$ tons/acre

LS = 0.326

C = 0.5

P = 1

From Equation 12, the dioxin concentration in sediment at Point 3 is:

$$C_{sediment} = C_{source}\left[\frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}}\right] \quad (12)$$

$$= 150 \text{ ug/Kg}\left[\frac{12 \text{ tons/acre/yr} \times 0.23 \text{ acre}}{10.5 \text{ tons/acre/yr} \times 448 \text{ acres}}\right]$$

$$= 150 \text{ ug/Kg}\left[\frac{2.76}{4700}\right]$$

$$= 0.088 \text{ ug/Kg}$$

Alternately, the basin size and estimated dioxin concentration at Point 2 can be used to estimate the dioxin concentration in sediment at Point 3, as follows:

$$C_{sediment} = C_{Point\ 2}\left[\frac{L_{Point\ 2} \times A_{Point\ 2}}{L_{basin} \times A_{basin}}\right]$$

$$= 1.8 \text{ ug/Kg}\left[\frac{7.35 \text{ tons/acre/yr} \times 32 \text{ acres}}{10.5 \text{ tons/acre/yr} \times 448 \text{ acres}}\right]$$

$$= 1.8 \text{ ug/Kg}\left[\frac{235}{4,700}\right]$$

$$= 0.09 \text{ ug/Kg}$$

APPENDIX B

EXAMPLE SITE 2

B.1 SITE DESCRIPTION

Consider the case of a private, rural lane (the source) that was sprayed several times with dioxin-contaminated oil to control dust. The lane is 0.25 mi (400 m) long and 15 ft (4.6 m) wide (0.45 acre). Sampling indicated that the average dioxin concentration in the source was 90 ug/Kg.

The source is located on a hill with a reported average slope of 3% (Figure B-1). A creek flows through the valley at the bottom of the hill. Soil type in the region is primarily silt loam. Average annual wind speed is 12 mph (5.4 m/sec), with the predominant wind direction down the valley.

B.2 PROBLEM

Sampling was conducted only in the source area where human health risks were considered to be highest. However, potential exposure to dioxin is also of interest for areas downwind, downslope, and downstream of the site. Concentrations of dioxin at such points of exposure can be predicted using the appropriate conversion factors for the various modes of environmental transport.

In this example, there are concerns about dioxin concentrations at the following locations:

1) atmospheric concentrations at the source;

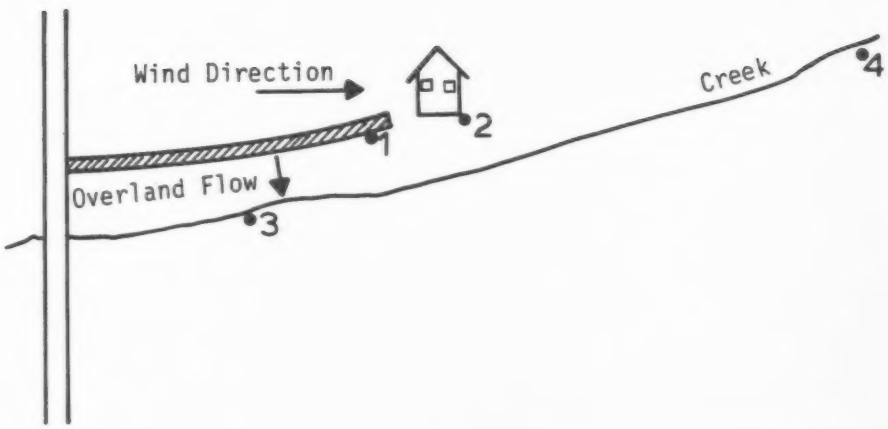

Figure B-1. Example Site 2

2) atmospheric concentrations near the residence, which is 100 ft (30 m) beyond the end of the lane in the downwind direction;

3) soil and atmospheric concentrations adjacent to the creek that is 500 ft (150 m) downslope from the source; and

4) sediment concentrations in the creek at a point 4,000 ft (1,200 m) downstream of Point 3.

The drainage basin at Point 3 has an average slope of 3%, an average slope length of 1,000 ft (300 m), and an area of 0.25 sq mi (160 acres).

The drainage basin at Point 4 has an average slope of 3%, an average slope length of 1,000 ft (300 m), and an area of 1 sq mi (640 acres).

Soil type and average annual wind speed at the points of interest are the same as those at the source.

B.3 ATMOSPHERIC CONCENTRATION AT THE SOURCE

Because the average wind speed is less than 10 m/sec, Equation 21 is used to calculate air concentrations.

$$CF_{air} = \frac{C_{air}}{C_{soil}} = 1 \times 10^{-7} \text{ Kg/m}^3 \qquad (21)$$

$$C_{air} = (1 \times 10^{-7} \text{ Kg/m}^3)\ C_{soil} \text{ ug/Kg}$$

$$C_{air} = (1 \times 10^{-7} \text{ Kg/m}^3)(90 \text{ ug/Kg})$$

$$C_{air} = 0.000009 \text{ ug/m}^3$$

$$= 9 \text{ pg/m}^3$$

B.4 ATMOSPHERIC CONCENTRATION AT THE RESIDENCE

Equation 21 is modified to include the dispersion factor for calculating air concentrations downwind of the source.

$$CF_{air} = \frac{C_{air}}{C_{soil}} = 1 \times 10^{-7}\left[\frac{(\sigma_y\sigma_z)_F}{(\sigma_y\sigma_z)_{x+F}}\right] \quad (8)$$

where $(\sigma_y\sigma_z)_F = 460\ m^2$

$(\sigma_y\sigma_z)_{x+F} = 540\ m^2$

$k_p = 0.001\ sec^{-1}$

$x = 30\ m$

$F = 400\ m$

$W = 5.4\ m/sec$

Substituting these values into Equation 8:

$$\frac{C_{air}}{C_{soil}} = (1 \times 10^{-7})\left[\frac{460}{540}\right]$$

$$C_{air} = 0.00000009\ ug/m^3$$

$$= 0.09\ pg/m^3$$

B.5 SOIL CONCENTRATION NEAR THE CREEK

The soil lost from the source is

$$L = R \times K \times LS \times C \times P\ tons/acre/yr = 20\ tons/acre/yr$$

where $R = 215\ yr^{-1}$

$K = 0.4\ tons/acre$

$LS = 0.233$

C = 1

P = 1

Because the source area is 0.45 acre, the mass of soil from the site is 9 tons/yr (8,200 Kg/yr). At a soil dioxin concentration of 90 ug/Kg, 738 mg of dioxin are transported from the source annually.

The soil lost from the drainage basin above Point 3 is:

$$L = R \times K \times LS \times C \times P \text{ tons/acre/yr} = 25 \text{ tons/acre/yr}$$

where $R = 215 \text{ yr}^{-1}$

$K = 0.4$ tons/acre

$LS = 0.573$

$C = 0.5$

$P = 1$

With a drainage basin of 160 acres, the mass of soil lost from the basin is 4,000 tons/yr (3,629,000 Kg/yr).

From Equation 9, the dioxin concentration in soil at Point 3 is:

$$C_{soil} = C_{source} \left[\frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} \right] \qquad (9)$$

$$= 90 \text{ ug/Kg} \left[\frac{20 \times 0.45}{25 \times 160} \right]$$

$$= 0.2 \text{ ug/Kg}$$

Because the predominant wind direction is down the valley (at right angles to overland flow direction), atmospheric deposition need not be considered in calculating soil concentrations at Point 3.

B.6 ATMOSPHERIC CONCENTRATION NEAR THE CREEK

Because the predominant wind direction is down the valley (at right angles to overland flow direction), dioxin in the soil at Point 3 will be the primary source of atmospheric contaminants. At an average wind velocity of 5.4 m/sec, atmospheric concentrations will be due to vaporization, according to Equation 21:

$$C_{air} = (1 \times 10^{-7}\ Kg/m^3) C_{soil}\ ug/Kg \qquad (21)$$

$$= (1 \times 10^{-7}\ Kg/m^3)(0.2\ ug/Kg)$$

$$= 0.00000002\ ug/m^3$$

$$= 0.02\ pg/m^3$$

B.7 SEDIMENT CONCENTRATION DOWNSTREAM

Soil losses from the source are the same as those calculated in Section B.5. Soil losses from the basin upstream of Point 4 are:

$$L = R \times K \times LS \times C \times P = 25\ tons/acre/yr$$

where $R = 215\ yr^{-1}$

$K = 0.4$ tons/acre

$LS = 0.573$

$C = 0.5$

$P = 1$

From Equation 12, the dioxin concentration at Point 4 is:

$$C_{sediment} = C_{source} \left[\frac{L_{source} \times A_{source}}{L_{basin} \times A_{basin}} \right] \qquad (12)$$

$$= 90 \text{ ug/Kg} \left[\frac{20 \times 0.45}{25 \times 640} \right]$$

$$= 0.05 \text{ ug/Kg}$$

Alternately, the basin size and estimated dioxin concentration at Point 3 can be used to estimate the dioxin concentration in sediment at Point 4, as follows:

$$C_{Sediment} = C_{Point\ 3} \left[\frac{L_{Point\ 3} \times A_{Point\ 3}}{L_{basin} \times A_{basin}} \right]$$

$$= 0.2 \text{ ug/Kg} \left[\frac{25 \text{ tons/acre/yr} \times 160 \text{ acres}}{25 \text{ tons/acre/yr} \times 640 \text{ acres}} \right]$$

$$= 0.2 \text{ ug/Kg} \left[\frac{4{,}000}{16{,}000} \right]$$

$$= 0.05 \text{ ug/Kg}$$